A limnological study of selected lakes in the Lac de Gras area, Northwest Territories with special reference to fish contaminants

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ABSTRACT

Martin, K.A. 2001. A limnological study of selected lakes in the Lac de Gras area, Northwest Territories with special reference to fish contaminants. Can. Tech. Rep. Fish. Aquat. Sci. 2385:viii + 78 p.

Baseline data for fish contaminants are provided for fish from lakes in the BHP diamond mining claim block north of Lac de Gras, NT. At the time of this study there were no apparent differences in the contaminant levels in fish from the two drainage basins sampled. Fish had been exposed to a variety of pollutants, however levels were generally in a range comparable to other lakes in the Northwest Territories and would be considered fairly low. Results from metals analysis do indicate levels of Se, Cu and Hg in some fish high enough to be of concern. Continued monitoring of the fish from lakes in the area particularly for metal analysis is warranted.

Chironomidae emergence was sampled in both lakes, as was the zooplankton community structure. Both are important in the diet of all fish in the lakes. Both lakes have populations of lake trout (*Salvelinus namaycush*), round whitefish (*Prosopium cylindraceum*), arctic grayling (*Thymallus arcticus*), burbot (*Lota lota*), Cyprinidae, and sculpin, although data were only collected on the first three species. Round whitefish dominated the fish community in both lakes. Fish2 Lake supports a larger population of lake trout than arctic grayling, which is the opposite of the community structure in Fish1 Lake. Both round whitefish and arctic grayling populations were dominated by smaller size classes and lake trout by larger ones. Within a given lake, arctic grayling were positioned at the lowest trophic level and lake trout at the highest with round whitefish intermediate between them. Streams were briefly surveyed and were used by arctic grayling, lake trout, burbot, and sculpin, and allowed access between the two lakes.

Key Words: Lac de Gras; diamond mining; contaminants; lake trout; arctic grayling; round whitefish; zooplankton; limnology; metals.

RÉSUMÉ

Martin, K.A. 2001. A limnological study of selected lakes in the Lac de Gras area, Northwest Territories with special reference to fish contaminants. Can. Tech. Rep. Fish. Aquat. Sci. 2385:viii + 78 p.

Les données de base sur la présence de contaminants sont fournies pour les poissons des lacs de la concession de l'exploitation minière de diamants BHP, au nord du lac de Gras (T.N.-O.). Au moment de l'étude, il ne semblait pas y avoir de différences apparentes dans les concentrations de contaminants chez les poissons des deux bassins hydrographiques échantillonnés. Les poissons avaient été exposés à toute une gamme de polluants, mais les niveaux se situaient généralement dans une plage comparable à celle des autres lacs des T.N.-O. et peuvent être considérés comme assez faibles. Les résultats des analyses des métaux indiquent cependant chez certains poissons des concentrations de Se, Cu et Hg assez élevées pour susciter des préoccupations. La poursuite de la surveillance des poissons des lacs de la région se justifie donc, particulièrement en ce qui concerne les métaux.

L'émergence des chironomidés a donné lieu à un échantillonnage dans les deux lacs, tout comme la structure de la communauté zooplanctonique. Il s'agit là de deux ressources importantes dans le régime alimentaire de tous les poissons des lacs. Les deux lacs abritent des populations de touladi (*Salvelinus namaycush*), de ménonimi rond (*Prosopium cylindraceum*), d'ombre arctique (*Thymallus arcticus*), de lotte (*Lota lota*), de cyprinidés et de chabots, mais on a recueilli des données sur les trois premières espèces seulement. Le ménonimi rond dominait la communauté ichtyenne dans les deux lacs. Le lac Fish2 abrite une plus grande population de touladi que d'ombre arctique; c'est le contraire dans la structure de la communauté du lac Fish1. Les populations de ménomini rond et d'ombre arctique étaient dominées par les classes de petite taille, et la population de touladi par des classes de plus grande taille. Dans un lac donné, l'ombre arctique se positionnait au niveau trophique le plus bas, et le touladi au niveau le plus haut, le ménomini rond occupant une position intermédiaire. Nous avons procédé à une brève reconnaissance des cours d'eau, ils étaient fréquentés par l'ombre arctique, le touladi, la lotte et les chabots, et constituaient un passage entre les deux lacs.

Mots clés : Lac de Gras; l'exploitation minière de diamants; contaminants; touladi; ombre arctique; ménomini rond; zooplancton; limnologie; métaux.

INTRODUCTION

Most fisheries studies carried out in the Canadian Arctic limit themselves to either contaminant monitoring (Lockhart et al. 1989; Sanderson et al.1997) or fish population studies (Parker and Johnson 1991). Few limnological and biological resource analysis studies have been carried out in the arctic (Rigler 1972, 1975, Welch 1985). Usually the approach taken is similar to that of Environment Canada's Land Use Information Series from the 1970's and 1980's, which provided point-in-time limnological and biological information from a wide area of the arctic. Recently, research is moving into the combination of these areas (Boyle et al. 1992) in order to provide a better baseline from which to monitor any changes to the environment and to better explain some of the current findings.

Exploratory work by BHP Minerals Canada Ltd. (Vancouver, BC) related to diamond mining had been ongoing in the Lac de Gras area since 1990 (Rescan 1993a). Diamonds are associated with kimberlite, a type of volcanic rock, which is found in this area. Most kimberlite pipes are found at the bottom of lakes, as the lakes had formed in the glacier eroded crater surface of these pipes. Exploratory work begins with geophysical surveys usually carried out from the air. Geochemical sampling of promising sites is then undertaken to test for indicator minerals, which involves removing small samples of material from around the lake. If the samples indicate the potential for diamond bearing kimberlite, small diameter test coring is either done from shore at a 45° angle to intersect the target or it is done from the ice surface in the case of larger lakes. Small diameter cores are taken and analyzed for indicator minerals. When these tests reveal potentially suitable deposits, bulk sampling (large diameter reverse circulation drilling) of material from the kimberlite pipe is undertaken to remove sufficient material for processing using large diameter reverse circulation drills. Bulk sampling may involve anywhere from 20 to 200 tons of material. Exploratory work, principally bulk sampling and ultimately mining operations have had and will continue to have a significant impact on lakes in the area. For example, bulk sampling was carried out to determine the quantity and quality of the diamond resources on Koala and Fox lakes in 1993/1994 (Rescan 1993a). Bulk sampling occurred in Grizzly Lake in 1993 (D. Dyck, BHP Diamonds Inc., 8-1699 Powick Rd., Kelowna, BC, V1X 4L1, pers. comm.). Preliminary drilling was carried out on Sable Lake north of the Koala Watershed in1995 and 1996, and Misery Lake, southeast of the South Watershed in 1995 (Rescan 1996). To proceed with full-scale mining development, the lakes must first be drained. Koala Lake was drained and open pit mining began in 1995. A number of other lakes in the area are scheduled for drainage and mine development over the next 20 to 30 years including Panda, Leslie, Fox1 and Misery lakes (Rescan 1995). In addition, other lakes are to be used for tailings disposal (Long, Willy, Brandy, and Nancy lakes) (Rescan 1995). Airstrip Lake was drained to facilitate aggregate extraction and five small lakes will be used to dump waste rock (West Panda, North Leslie, South Fox, Fox 2 and North Misery lakes) (Rescan 1995). In addition, numerous (approximately 45) connecting, inflow and outflow streams associated with the lakes will be affected. Changes have been proposed for watersheds in the area such as a diversion channel to reconnect Panda Lake and Kodiak Lake following the draining of Koala Lake (Rescan 1994b). Other lakes are the sources of water required for camp (Larry Lake) or mining operations (Little Lake). Along with the direct impact on the lakes from the exploratory and mining operations there was likely some impacts from the facility construction

phase of the project. The main Koala Campsite itself includes a sample storage building, processing plant, fuel storage areas, sewage treatment plant and residence complex. A plant-site was also developed near the mining operations. An all-weather road was built to allow access to many of the lakes of interest for development (Rescan 1993c). This road was built primarily along the eskers, which formed barriers between drainage areas. A 45-km winter road was built to allow access to the site during the winter (Rescan 1993a) and an airstrip capable of handling Hercules aircraft was built adjacent to the campsite to carry personnel and supplies into and out of the area (Rescan 1993a).

Survey work on the lakes and streams to document their physical and biological characteristics early in the project was commissioned by BHP (Rescan 1993c, 1995). Data have been collected on fish removed from lakes where water levels have been reduced: Panda in 1996, Koala in 1997, Long in 1997, and Airstrip in 1997, and there is a continuing requirement for BHP to monitor the lakes in the area.

Initiated in 1993, the Lac de Gras Project (originally identified as the Izok Lake Project) was categorized as inventory/monitoring and environmental analysis under the Sustainable Fisheries Program. The objectives of the original project were to provide baseline resource and contaminants information from undisturbed lakes in the mineralogically important Slave Geological Province by describing limnological and biological variables. Enhanced knowledge of the resources in an area that was becoming the focus of intensive exploration, as well as potential and actual development was considered to be important. It should be noted that although the intention was to describe undisturbed lakes, the area had been subjected to exploratory work since 1990 and some disturbance may have resulted from this. The study involved monitoring limnological variables twice weekly (phytoplankton chlorophyll, nutrients, and thermal structure), quantification of benthic insect emergence (a measure of benthos production), and zooplankton biomass and species composition. Fish sampling was undertaken to provide population estimates and tissue samples for contaminant analysis.

METHODS

STUDY AREA

Field sampling began early in 1994 on lakes in the Koala Camp area of BHP Minerals Canada Ltd.'s Diamond Project on the north shore of Lac de Gras, NT (Fig. 1). The project area is located 300 kilometres NNE of Yellowknife, NT and BHP's original "Koala" camp is located at 64° 41' 18.0" N, 110° 36' 59.0" W (Fig. 1). Locations of all lakes sampled and a summary of morphometric characteristics are included in Table 1. In the summer of 1994, two lakes (Fish1 and Fish2) in the South watershed area, currently in a drainage where no kimberlite has been discovered, were selected for intensive study for the three summers 1994-96 (Appendix 1). They were considered relatively undisturbed in comparison to many other lakes in the immediate vicinity. They were chosen based on their size, accessibility, and ultimately because the bedrock geology beneath each study lake is non-kimberlitic, and therefore not of interest for mine development. A laboratory facility was provided at the Koala Camp where sample

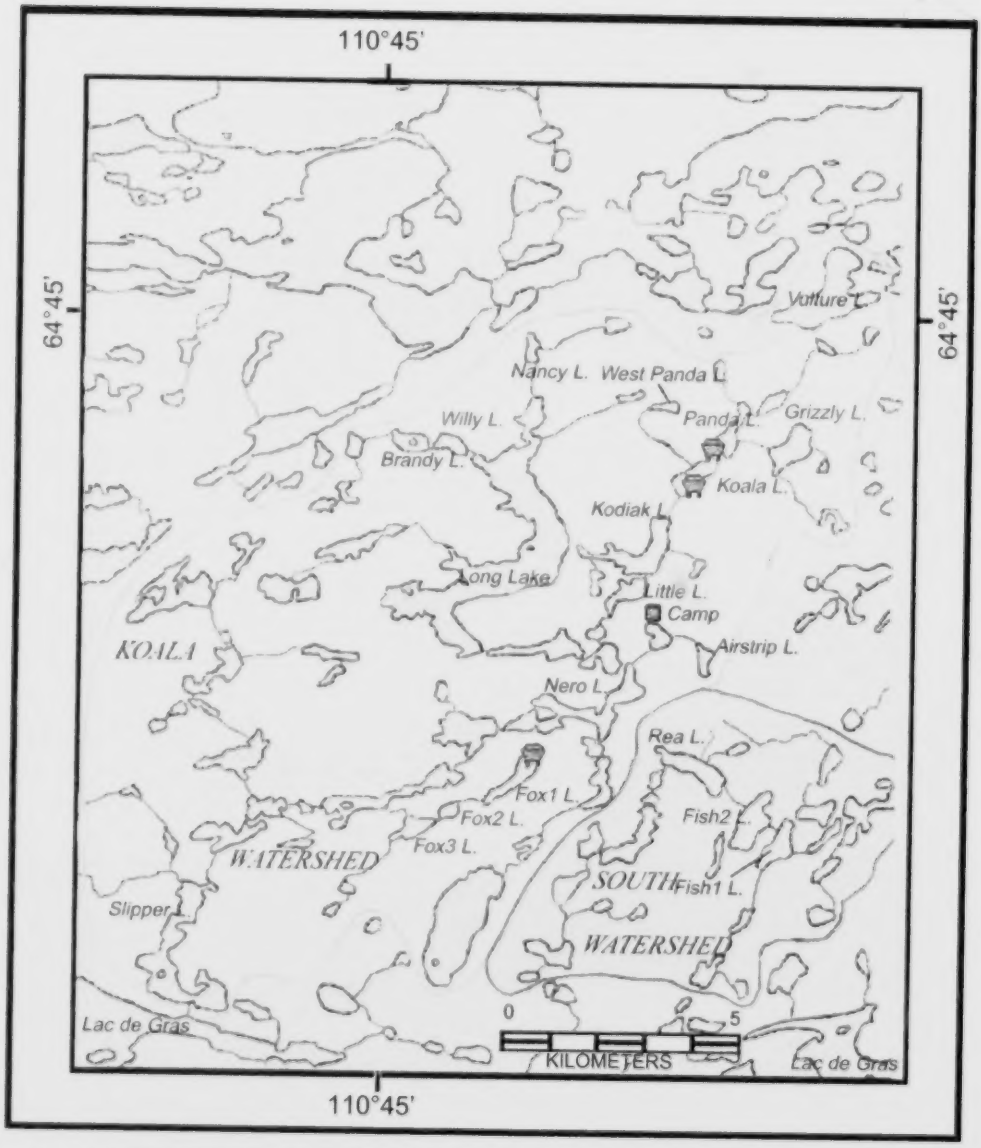

Fig. 1. Lakes sampled as part of this study as well as lakes affected by exploratory and mining development in the Lac de Gras area. The original Koala campsite and the first full-scale mining operations are identified.

Table 1. Lake morphometry characteristics for the lakes sampled from 1994 to 1996. Buster Lake drained into Kodiak Lake (north of the Panda Diversion Channel) and Lac de Gras was sampled at the inlet near Slipper Lake. Nd indicates no data were available. All data summarized from Rescan (1993b).

Lake Name	Location	Surface Area (ha)	Max. Depth (m)	Mean Depth (m)	Cumulative Volume (10^4 m^3)
Fish ecology (*and contaminant sampling)					
Fish1 (Cam Lake)*	64° 38' 27.0" N, 110° 33' 22.9" W	45.2	6	1.88	84.9
Fish2 (Ron Lake)*	64° 38' 43.3" N, 110° 34' 15.4" W	68.0	18	3.92	266.4
Rea Lake	64° 39' 30.0" N, 110° 36' W	nd	nd	nd	nd
Contaminant Sampling (**and oxygen measurements):					
Fox1 Lake	64° 39' 16.6" N, 110° 41' 9.5" W	43.7	29	6.9	303.0
Grizzly Lake**	64° 43' 29" N, 110° 33' 07" W	58.9	43	14.6	857.9
Kodiak Lake**	64° 42' 16.9" N, 110° 37' 5.7" W	90.7	13	2.1	193.8
Lac de Gras	64° 38' 50" N, 110° 34' 23" W	nd	nd	nd	nd
Long Lake**	64° 42' 6.2" N, 110° 40' 48.6" W	614.4	32	7.4	4528.7
Nero Lake**	64° 40' 7.9" N, 110° 38' 23" W	133.6	16	2.8	373.1
Buster Lake	64° 42' 26.2" N, 110° 34' 11.9" W	nd	nd	nd	nd
Vulture Lake	64° 45' 05" N, 110° 33' 12" W	180.2	43	11.2	2025.9

preparation (pre-weighing containers, drying samples, etc.) and storage of samples was carried out.

Fish1 Lake is a shallow single basin lake (45.2 ha, max depth 6.0 m)(Fig. 2). A large volume (73%) of the lake is less than 2 m deep, much of which would be frozen during the winter. Fish2 Lake is larger and deeper (68.0 ha, max depth 18.0 m, 40% of the volume < 2 m deep)(Fig. 3). Secchi depths varied throughout the summer and averaged 3.5 m in Fish1 Lake and 5.7 m in Fish2 Lake. Water flows into Fish2 Lake from the northwest and flows out the most easterly point through a small stream into Fish1 Lake. Fish1 Lake has a second inflow from the south end and water flows out of the lake from the most easterly and most northerly points. There are several deep spots and very steep drop-offs along the western side of Fish2 Lake. In both lakes the shallow water (≤ 2 m) was rocky and the area below two metres was finer substrate. Several spots at the south end of Fish1 Lake had organic material in the shallow water. Fish2 Lake had an extensive sandy shoreline along its north and northeastern shores. It also had extensive areas of aquatic vegetation along its shores not present in Fish1 Lake.

WATER SAMPLING

In 1994, winter oxygen readings were measured in several lakes using the methodology described in Welch and Bergmann (1985). Ice thickness, at the sampling hole, was measured using a marked stick with a right-angle bracket secured to the bottom. Average snow depth was determined from 17 measurements taken at 4 m intervals radiating from the sampling hole using a metre stick. Beginning in 1994, Fish1 and Fish2 lakes were usually sampled at weekly intervals during July and August each year at the deepest spot in the lake (centre marker). Water samples were collected using a 2L Van Dorn type sampler. In 1994 samples were taken at 1 m depth. In 1995 and 1996 samples were taken from two depths: 1 m and 4 m both years in Fish1 Lake, and 1 m and 10 m in 1995 and 1 m and 4 m in 1996 in Fish2 Lake. Samples were prepared and sent to the Freshwater Institute's Analytical Chemistry Laboratory in Winnipeg for analysis as per Stainton et al. (1977). Ignited GF/C filters used to filter 500 mL water were frozen, shipped to Winnipeg, and then analyzed to determine the concentration of suspended carbon (Susp C), suspended nitrogen (Susp N), suspended phosphorus (Susp P), suspended iron (Susp Fe), and chlorophyll *a* (Chl). Samples for Chl analysis (1995 and 1996 only) were handled under dim light and the filters were stored in vials wrapped in aluminum foil prior to analysis. To determine the concentration of soluble reactive silica (SRSi), 500 mL of water was filtered through an ignited GF/C filter and the filtrate stored in plastic vials, refrigerated, and shipped to Winnipeg for analysis. Frozen samples of the same filtrate were also analyzed to determine the concentration of total dissolved nitrogen (TDN) and total dissolved phosphorus (TDP). Air and surface water temperatures were measured, temperature profiles recorded at one metre intervals, prevailing weather conditions noted, and Secchi depths were measured.

ZOOPLANKTON

Zooplankton samples were collected biweekly during July and August each year from the centre marker in Fish1 and Fish2 lakes. Samples were collected with vertical hauls using a 40 cm diameter 100µ mesh net (1256 cm^2 mouth opening) dropped to within 0.5 m of the bottom. One haul was concentrated and preserved with 10%

Fig. 2. Bathymetry of Fish1 Lake showing emergence trap transects. Arrows indicate inflow and outflows from the lake.

Fig. 3. Bathymetry of Fish2 Lake showing emergence trap transects. Arrows indicate inflow and outflows from the lake.

formalin for identification (1994 and 1995 only). Three hauls were combined, concentrated, and then poured in pre-weighed aluminum weigh boats (all years). These were placed in an oven at 60°C until dry (usually 45 min to 1.5 h) and then placed in a desiccator for 24 h and weighed.

INSECT EMERGENCE

Emerging insects were collected in 1995 (July and August) and 1996 (June to August) in an attempt to quantify the secondary production in the lakes. Each of the two sample lakes had 25 conical emergence traps (60 cm diameter, 0.2827 m^2) based on the design of Davies (1984) with internal funnels as in Welch et al. (1988). Traps at depths of 0.5 and 1.5 m had extra lead on the rim and were placed directly on the bottom. The remaining traps were suspended by rubber tubing from pieces of wood 2 x 4's anchored at the target depth. The top of the jar was positioned approximately 18" below the surface to avoid wave action. The number of emergence traps placed at each depth was proportional to the lake area at that depth. Fish1 Lake had seven traps placed at 0.5 m, ten at 1.5 m, three at 2.5 m, two at 3.5 m, two at 4.5 m, and one at 5.5 m. Fish 2 Lake had five traps placed at 0.5 m, five at 1.5 m, three at 2.5 m, two at 3.5, two at 4.5 m, two at 5.5 m, two at 6.5 m, and one at each of 7.5 m, 9.0 m, 11.5 m, and 15.0 m. The same placement of traps was repeated each year. Traps were checked every second day. Contents were sorted to order, counted, wet weight determined, and then dried (as with zooplankton bulk samples), and dry weight determined. Emergence traps were found to contain both whole (intact) Diptera and Diptera heads on occasion throughout both sampling years which was attributed to Coleoptera larvae entering the traps and eating the bodies of the insects. In both years the number of heads found in samples was recorded so that the number of emerging individuals was accurate. In 1995, the sample weights included the whole individuals plus the heads. In 1996 the heads were not included in the weight of the sample. In 1996 a number of collections were made of heads alone. These were dried and weighed to determine the mean head weight that was then used to correct for the head weights included in the total sample weights for the 1995 data. The average weight of the intact individuals was applied to the entire number of animals collected in each sample for both years.

FISH COLLECTION

Fish were collected by angling, trap netting, gill netting, and electrofishing. Several trap net designs, based on those of Beamish (1972), were used during the course of the study. Two nets were set in each lake and were deployed for most of July and August each year. All were set perpendicular to shore in 1-3 m of water. Multi-filament experimental gill nets (250'-450' long with 1-4½" mesh) were set out from shore and nets were checked from 10 minutes to two hours. Electro-fishing was used to collect fish from the stream entering Fish2 Lake and the Fish1-Fish2 connector stream in 1994. In 1994 any small fish captured by traps were released without notation. In 1995 numbers and approximate sizes were noted and in 1996 more detailed records of captures and sizes were recorded.

Initially fish were collected for contaminant determination. Once this was completed, the sampling protocol changed and each fish captured was anaesthetized with MS-222 (tricaine ®) or 2-phenoxyethanol and measured to the nearest mm (fork length for Salmonidae, total length for burbot, *Lota lota*). All fish were also weighed in the field using either a Sartorius Portable PT6 or an Ohaus Portable Plus balance. All

fish captured were marked using individual sequences of fin scars (Welch and Mills 1981) except the earliest captures in 1994 that were batch marked. A section from the base of the pectoral fin was removed from each fish when marked for aging purposes. The pectoral notch became part of the individual fish marking code. Once data were collected and the fish recovered from the anaesthesia, they were released back into the lake.

Whole stomach samples were removed from fish that did not recover from the anaesthetic. In 1996 stomach samples were taken from live fish under anaesthesia using forceps or stomach pumps (Sotiropoulos 1997). The Relative Importance Index, RI (George and Hadley 1979) of prey taxa were calculated for lake trout (*Salvelinus namaycush*), round whitefish (*Prosopium cylindraceum*), and arctic grayling (*Thymallus arcticus*) using the equation:

$$RI_i = 100 \cdot \frac{AI_i}{\sum_{i=1}^{n} (AI_i)}$$

Where AI_i = the absolute importance of the prey taxon i,
= % frequency of occurrence + % total numbers + % total mass;
n = the number of different food types
% frequency of occurrence = the percentage of all stomachs containing prey taxon i,
% total numbers = the number of each prey taxon expressed as a percentage of the total number of items in the stomach,
% total mass = the blotted wet weight of each prey taxon expressed as a percentage of the total stomach content weight.

Some of the fish collected for contaminant analysis in 1994 were also analyzed for stable isotopes $\delta^{15}N$ and $\delta^{13}C$ to determine trophic relationships. Skinless dorsal muscle tissue from individual lake trout, round whitefish and arctic grayling was prepared and analyzed as per Kidd et al. (1998).

FISH ABUNDANCE

Following the completion of contaminant sampling, the three species of Salmonidae, along with a few burbot, were marked and recaptured in 1994, 1995 and 1996 in an attempt to estimate biomass and populations for the two study lakes (Fish1 and Fish2 lakes). Abundance estimates ($\hat{N}_i$) were made using the Jolly-Seber "death only" model after removing all new recruits from the analyses (Jolly 1965; Seber 1982). To estimate the abundance for the final period (1996), the Jolly-Dickson "full model with constant survival" was used (Jolly 1982; Jolly and Dickson 1980). Estimates were made for each lake and for each species except burbot, and included only fish $\geq$ age 1 at the start of the study. Burbot marked using the fin clip method were not easily or reliably identified upon recapture and as a result of this, population estimates based on mark-recapture were not made for them. All estimates were derived using the programs of Arnason et al. (1998) release B.3.

FISH DIET

Lake trout, arctic grayling, and round whitefish stomach samples were analyzed in 1996. Samples from Fish1, Fish2, and Rea lakes were analyzed, and the results combined to describe a general pattern of fish diet (Sotiropoulos 1997). The Absolute Importance of each prey taxa from late June to August (Sotiropoulos 1997) has been combined and the RI calculated for the entire sampling season. Unidentified material in the stomachs, usually too well digested to be identified, was not included for these analyses.

FISH CONTAMINANT SAMPLING

Under-ice gillnetting was carried out to collect fish for contaminant analysis from January to March 1994. Fish were frozen whole and shipped to the Ecotoxicology Laboratory, Fisheries and Oceans Canada (DFO), Burlington, for organochlorine analysis. In July and early August 1994, ten fish for each species (lake trout, round whitefish, and arctic grayling) were sampled for contaminant analysis. Along with the two principal sampling lakes, contaminant sampling was carried out on fish from lakes in the adjacent Koala watershed in 1994 (Table 1). All were processed at the lake within a few minutes of death, and tissues were quick-frozen on dry ice. They were kept on dry ice prior to and during shipping to the destination laboratories. Processing involved weighing and measuring the fish, noting reproductive status, removing, bagging, and weighing the liver, extracting bile with a 1 cm^3 syringe, measuring the volume and freezing both the liver and the bile, and then removing and preserving the stomach, otoliths, and left pectoral fin. The left side of the fish was filleted and sectioned for liver mixed function oxygenase (MFO) enzyme activity and metals analysis without the skin. The right side of the fish was filleted (skin intact) for polyaromatic hydrocarbon (PAH) analysis.

Analytical Methods

In all analyses reported with the exception of PAH calculations, the detection limit was used for statistical analysis when values were reported below the detection limit within a sample group.

Cantest Metals Analysis

Muscle tissue from 13 lake trout from five lakes were sampled for metals analysis (results provided by Cantest Ltd., Vancouver BC). The ICP Analysis program consisted of elements and their associated calibration standard solutions. The elements included aluminum (Al), antimony (Sb), arsenic (As), barium (Ba), beryllium (Be), boron (B), cadmium (Cd), calcium (Ca), chromium (Cr), cobalt (Co), copper (Cu), Iron (Fe), lead (Pb), magnesium (Mg), manganese (Mn), molybdenum (Mo), nickel (Ni), potassium (K), silver (Ag), sodium (Na), strontium (Sr), tin (Sn), titanium (Ti), vanadium (V), zinc (Zn) and phosphorous (as PO_4). Frozen muscle tissue subsamples (~ 5-6 g wet weight muscle tissue) were prepared using Nitric-Peroxide Acid Digestion. Samples were digested with 7 mL Nitric Acid (trace metal grade) for four hours over low heat, then cooled, 5 mL hydrogen peroxide (Merck Analytical Grade) was added, cooled, and then the sample was diluted to 50 mL with ultra pure water. A minimum of three reagent blanks and the NCR number Tort-1 (lobster tissue) and Dolt-2 (dogfish liver) certified reference materials were digested and analyzed concurrently.

I.C.P. (plasma) analysis: Analysis was performed using a Jarrell Ash Model #975 Inductively Coupled Argon Plasma Spectrograph equipped with Minipuls 2 Peristaltic pump and an "all glass" MAK High Pressure Nebulizer. The instrument was calibrated using standard solutions prepared from pure metals or pure metal salts. Calibration was checked after every ten solutions (blanks, certified standards or sample solutions) were analyzed. The instrument was recalibrated if the deviation from the standard value was over two percent.

Mercury (Hg) was determined after the same digestion without pretreatment, using a Pharmacia Model 100 M Mercury monitor equipped with a Servo graphic recorder model 410. As, Cd, Pb and Ni were determined using a Varian Model Spectral 400 Zeeman Background Corrected Graphite Furnace. Cu, Zn, and crosscheck of high Cd values were determined using a Varian Model 475 Direct Flame Atomic Absorption spectrophotometer.

Cantest reported the results of their metal analyses in dry weight. Dry weight values for fish tissue analyses were converted to wet weight units

$$\text{Wet weight} = (\text{dry weight value}) \cdot (1-(\%\text{moisture}/100))$$

Wet weight detection limits were determined using the moisture content of each fish muscle sample (average = 79.5%). Moisture values reported were determined on separate aliquots after drying at 105°C overnight.

DFO Metals Analysis

Muscle and liver tissue samples from Lac de Gras area arctic grayling, lake trout, and round whitefish were analyzed for As, Cd, Cu, Pb, Hg, selenium (Se), and Zn by the Environmental Chemistry Laboratory, Freshwater Institute, DFO, Winnipeg. All results are reported on a wet weight basis. All acids were trace metal analysis grade (concentrated) unless otherwise specified. All water was distilled and deionized. Commercial atomic absorption standards, reagent blanks, and certified reference material (NRC Dorm-1 and Dolt-2) were carried through the entire procedure. Test tubes used for tissue digestion were 25 x 200 mm pyrex glass and were washed with 10% nitric acid, followed by multiple rinses with distilled, deionized water prior to use. A time and temperature programmable aluminum block test tube heater was used for the digestion.

Hg was analyzed using the Hot Block Digestion – Cold Vapour Atomic Absorption Method (Hendzel and Jamieson 1976). A sub-sample of wet (liver/muscle) tissue (0.2 g) was digested with 5 mL of 4:1 sulfuric/nitric acids at 180°C for 12 hours, cooled, and diluted to 25 mL with water. Elemental Hg was released from this solution with stannous chloride reductant and carried by a stream of air to a LDC model 3200 Hg Monitor for cold vapour atomic absorption detection.

As and Se were analyzed using the Borohydride Reduction Method (Vijan and Wood 1974). A sub-sample of wet tissue (0.8 g muscle, 0.2 g liver) was digested with 4 mL nitric, 0.5 mL sulfuric and 1 mL perchloric acids for five hours at 130°C, followed by two hours at 200°C. After addition of 15 mL water and 7.5 mL hydrochloric acid, the solution was heated to 90°C for one hour, cooled, and adjusted to 25 mL with water. Arsine gas was generated from this solution by the automated addition of 2% sodium borohydride and 10% potassium iodide solutions, and swept by a nitrogen gas stream into an electrically heated quartz tube furnace (800°C) installed in the burner cavity of a

Varian SpectrAA-20 Atomic Absorption Spectrophotometer. Similarly, selenium hydride was generated by the automated addition of 2% sodium borohydride, followed by detection in the manner described above.

For Cu and Zn, a sub-sample of wet tissue (5 g muscle, 2 g liver) was digested with 5 mL nitric, 0.5 mL sulfuric and 2 mL of perchloric acids for 5 hours at 130°C, followed by two hours at 200°C. After cooling, the sample was adjusted to 25 mL with water and analyzed by air-acetylene flame atomic absorption (Varian SpectrAA-20) with deuterium background correction off for Cu and on for Zn analysis.

For Cd and Pb analysis sample digestion was similar to that described for Cu and Zn, with the exception that the sulfuric acid component was reduced to 0.2 mL. Solutions were analyzed by Zeeman background corrected, graphite furnace atomic absorption spectrophotometry (Hitachi Polarized Zeeman Model Z8200).

Five lake trout muscle samples were analyzed by both laboratories for As, Cd, Cu, Pb, Hg, and Zn.

Liver Mixed Function Oxygenase (MFO) Enzyme Activity

Liver samples from arctic grayling, lake trout, and round whitefish were collected for MFO analysis from seven lakes. Analyses were conducted by the Contaminants Laboratory, DFO Freshwater Institute Laboratory, Winnipeg. All results are reported on a wet weight basis. Liver samples were analyzed for liver enzymatic activity assays for the following three MFOs: aryl hydrocarbon hydroxylase (AHH), ethoxyresorufin-*O*-deethylase (EROD), and cytochrome P-450 (P450).

Two mixed function oxygenase enzyme activities were measured in microsomes prepared from the liver samples. The AHH assay was based on that of Depierre et al. (1975), as modified by Van Cantfort et al. (1977). Three separate incubations were carried out for each microsomal suspension, and in each case triplicate blanks were run. EROD was measured by the deethylation of 7-ethoxyresorufin to yield resorufin, which was detected by the fluorometric procedure described by Pohl and Fouts (1980). All EROD results were expressed in terms of the more recent resorufin standards, which are of higher purity. Blanks, consisting of samples to which the methanol had been added prior to the addition of substrate, were run for each sample, and the final result corrected for any non-enzymatic production of resofurin. Triplicate incubation mixtures of each sample were run, along with triplicate blanks for each sample. The detection limits for EROD and AHH were 0.001 ηmol· mg protein^{-1}·min^{-}1.

The procedure for measuring cytochrome P-450 was based on methods described by Omura and Sato (1964a and b). Analyses were done in triplicate. The detection limit was 0.001 ηmol ·mg·protein^{-1}.

Organochlorine (OC) Contaminants

Analyses were conducted by the Ecotoxicology Laboratory, DFO, Burlington. All results are reported on a wet weight basis. Organochlorine (OC) analysis of the whole fish samples included α-, β-, γ-, and δ-HCHs, octachlorostyrene, α/cis-chlordane, γ/trans-chlordane, dieldrin, aldrin, *p,p'* DDE, *p,p'* DDD, *o,p'* DDT, *p,p'* DDT, ΣDDT, HCB, heptachlor, heptachlor epoxide, endrin, β-endosulfan, methoxychlor, mirex, photomirex. ΣDDT includes the total of *p,p'* DDE, *p,p'* DDD, *o,p'* DDT, *p,p'* DDT, and *o,p'* DDE. Detection limits reported are the method detection limits. Samples were blended with sodium sulphate, packed into a column, and eluted with methylene chloride. Lipids

chloride. Lipids were then removed from the extracts via GPC. After extraction, samples were further cleaned up on silica gel columns. Analysis and quantification were performed by high resolution GC/ECD. Results were corrected for surrogate recoveries. Further analysis methods are included in Huestis et al. (1995).

Toxaphene

Analyses were conducted by the Ecotoxicology Laboratory, DFO, Burlington. All results are reported on a wet weight basis. Approximately five grams of tissue was ground with Na_2SO_4 and extracted with methylene chloride. Lipids were removed by automated GPC. Extracts were further cleaned up and fractionated on a 1% deactivated silica gel column eluted with hexane (A), followed by 1% methylene chloride in hexane (B). Fraction A contained PCBs and some OCs, while fraction B contained Toxaphene and the majority of the OCs.

Toxaphene was determined using a Varian 3600 gas chromatograph (GC) equipped with dual electron capture detectors (ECD) using 60m Restek RT_x5 and 60m Restek R_xt 1701 capillary columns. One µL of sample was injected directly onto the column, with the injector and detector temperatures at 240°C and 300°C respectively. The temperature program was: hold at 80°C for one min; ramp to 180°C at 15°C·min^{-1}; ramp to 260° C at 2°C·min^{-1}; hold for five min; ramp to 270°C at 5°C·min^{-1}, and hold for 17 min. Quantification of toxaphene was based on a set of 20 peaks with a method detection limit of 30 ng·g^{-1}. Total toxaphene was quantified by summing the area counts of 20 potential target peaks in the sample and comparing this to the sum of the same peaks from the technical toxaphene standard (Hercules). Results were corrected for surrogate recoveries. Further analytical methods are included in Huestis et al. (1995).

Polychlorinated Biphenyls (PCBs)

Analyses were conducted by the Ecotoxicology Laboratory, DFO, Burlington. All results are reported on a wet weight basis. Whole fish samples were analyzed for total PCBs, a measurement of the concentration of the sum of all peaks found in the sample quantified against a 1:1:1 analytical mixture of Aroclors 1242, 1254, and 1260 (Huestis et al. 1995). The concentrations of the following (54) single congeners of PCB: 15, 18, 28, 31, 32, 40, 42, 44, 47/48/75, 49, 50, 52, 56/60, 64, 66, 70, 74, 76, 84, 85, 87, 89/101, 97, 99, 105*, 110, 118, 128, 129, 136, 137, 138, 141, 149, 151, 153, 156, 158, 170, 171, 172, 177, 178, 180, 183, 185, 187, 193, 194, 195, 196/203, 199, 201, and 206 were also determined. Coplanar (non-ortho) PCBs 77, 81, 126 and 169 were also analyzed. At the time of analysis PCB 105 was considered a coplanar PCB so the data were reported with the coplanars but it is no longer considered to be a coplanar and is therefore included in the list of PCB isomers which the samples were analyzed for. Samples were processed as described in the previous section on toxaphene. After silica gel column fractionation, samples were analyzed by a combination of high resolution GC/ECD to determine the original list of congeners. The expanded list of isomer specific PCBs were measured by GC/MS detection. Coplanar (non-ortho) PCB samples were cleaned up on an alumina column, and separated from PCDDs and PCDFs by carbon chromatography on a high pressure liquid chromatographic system. Analysis and quantification were performed by high resolution GC/MS-SIM. Detection limits were in the low parts per trillion range for the coplanar PCBs, and parts per billion for the others. Results were corrected for surrogate recoveries. Further analytical methods are included in Huestis et al. (1995).

Dioxin and Furan Analyses

Analyses were conducted by the Ecotoxicology Laboratory, DFO, Burlington. All results are reported on a wet weight basis. Individual whole fish samples were analyzed for sixteen 2,3,7,8-substituted PCDD/Fs at the DFO Laboratory in Burlington. In addition, for each group of congeners (tetra, penta, hexa and hepta), the sum was calculated. Unlike homologue groups, these sum values include both 2,3,7,8-substituted and non-substituted congeners. Samples were blended with sodium sulphate, packed into a column, and eluted with methylene chloride. Lipids were removed from the extracts via GPC. After extraction, samples were further cleaned up on an alumina column, and fractionated by carbon chromatography on a high-pressure liquid chromatographic system into two extracts, one containing coplanar PCBs, and the other containing PCDDs and PCDFs. This carbon fractionation also cleaned the PCDD/F extract of other planar contaminants, including diphenyl ether, which can cause interference at the TCDF signal (Huestis and Sergeant 1992). Analysis and quantification were performed by high resolution GC/MS. Spike recoveries were calculated using radio-labeled internal standards. Results were corrected for surrogate recoveries. Further analytical methods are included in Huestis et al. (1995).

Polyaromatic Hydrocarbons (PAH)

Analyses were conducted by the Contaminants Laboratory, DFO, Freshwater Institute, Winnipeg. All results are reported on a wet weight basis. The PAHs in fish tissues were extracted by the methanol-potassium hydroxide reflux method. The procedure involved digestion, solvent extraction, and column chromatography. A few mL of bile and approximately 10 g of muscle were required. The extracts were analyzed using gas chromatography/mass selective detection (GC/MSD). Lockhart et al. (1989) provides a detailed description of the method used. PAH metabolites were not analyzed in the bile samples.

RESULTS AND DISCUSSION

WINTER MEASUREMENTS

Winter measurements of dissolved oxygen, snow depth, and ice thickness, were made on several Lac de Gras area lakes at the start of the study (Table 2). Maximum ice thickness measured in 1994 was 1.875 m on Grizzly Lake and mean snow depth ranged from 10.9 cm to 24.8 cm over all the lakes. Lowest levels of dissolved oxygen were found in Kodiak Lake and ranged from a minimum of 2.65 ppm to a maximum 10.52 ppm (mean value 4.92 ppm) on 26 March. Bulk sampling had been carried out in Grizzly Lake in 1993 and was being carried out in Koala Lake (upstream of Kodiak Lake) at the time of the winter measurements (D. Dyck, pers. comm.). There was no work carried out on Kodiak Lake itself prior to or during the sampling for winter Oxygen (D. Dyck, pers. comm.). It was expected that this would be near the lowest oxygen levels experienced prior to the onset of primary production triggered by the increasing spring light input through the ice. There was no attempt to determine any influence of drilling activity on the winter oxygen levels as part of this study.

Table 2. Dissolved oxygen, snow depth, and ice thickness measurements from early 1994.

Lake	Measurement	Sample 1	Sample 2	Sample 3	Sample 4
Grizzly Lake	Sample Date	7 Feb 1994	24 Feb 1994	9 March 1994	26 March 1994
	Max O_2 (ppm)	14.21	14.89	14.89	14.77
	Min O_2 (ppm)	8.75	10.43	8.85	7.54
	Mean O_2 (ppm)	12.04	13.01	12.76	12.45
	Mean Snow depth(cm)	10.9	15.1	20.7	17.1
	Ice Thickness (m)	1.48	1.71	1.79	1.87
Kodiak Lake	Sample Date	24 Feb 1994	9 March 1994	26 March 1994	
	Max O_2 (ppm)	12.14	12.26	10.52	
	Min O_2 (ppm)	3.08	3.61	2.65	
	Mean O_2 (ppm)	6.22	6.03	4.93	
	Mean Snow depth(cm)	20.4	23.0	24.8	
	Ice Thickness (m)	1.37	1.29	1.52	
Long Lake	Sample Date	6 Feb 1994	5 March 1994	25 March 1994	
	Max O_2 (ppm)	12.65	14.28	13.82	
	Min O_2 (ppm)	6.66	5.32	2.87	
	Mean O_2 (ppm)	11.48	11.72	10.75	
	Mean Snow depth(cm)	12.4	21.0	20.7	
	Ice Thickness (m)	1.19	1.43	1.60	
Nero Lake	Sample Date	4 Feb 1994	5 March 1994	25 March 1994	
	Max O_2 (ppm)	15.79	16.12	15.36	
	Min O_2 (ppm)	5.14	6.14	6.10	
	Mean O_2 (ppm)	11.15	11.64	11.39	
	Mean Snow depth(cm)	11.7	14.7	15.0	
	Ice Thickness (m)	1.17	1.51	1.53	

TEMPERATURE

In early July 1994, when fieldwork began, the ice was already gone from the lakes. In 1995 and 1996, ice was completely off the lakes in the third week of June. Surface water temperatures were recorded over the three summers and indicate yearly differences (Fig. 4). In 1994, both lakes remained well mixed following ice out with surface temperatures reaching a maximum of 16.2°C in Fish1 Lake and 15.9°C in Fish2 Lake by late July and then gradually cooling off through August (Fig. 5 and 6). In 1995, July air temperature was cooler and water temperatures didn't warm up as quickly (Fig. 5 and 6). By mid-August, temperatures at the surface had reached 14 -15°C but temperatures were from 3 - 4.5°C lower at the bottom in both lakes (Fig. 5 and 6). The thermocline was measured between 4 and 4.5 m in Fish1 Lake (15 August 1995), and between 7.5 and 8 m in Fish2 Lake (18 August 1995), but by 20 August Fish2 Lake water was completely mixed again. In 1996 high air temperatures (above 20°C) in late June resulted in water warming quickly (Fig. 5 and 6). By 4 July 1996 surface water had reached 17.5°C in Fish1 (16°C Fish2 Lake). Fish1 Lake was stratified for a few days only. Fish2 Lake remained stratified throughout July, only mixing again in early August. The temporary thermal stratification is similar to that found in the Saqvaqjuac lakes of northern Hudson Bay, NU (Welch 1985).

NUTRIENTS AND CHLOROPHYLL

Concentrations of nutrients and chlorophyll were measured over the three years (Table 3) and in all cases, except Si and Chl, the concentrations were slightly higher in Fish1 Lake than in Fish2 Lake (Table 3). Total P (sum of TDP and Susp P) summer open water concentrations ranged from 4 - 16 $\mu g \cdot L^{-1}$. Values from the Saqvaqjuac area lakes ranged from 5 to 10 $\mu g \cdot L^{-1}$ for prefertilization levels and reached 10 - 20 $\mu g \cdot L^{-1}$ after fertilization (Welch et al. 1989). Total N (sum of TDN and Susp N) ranged from 99 - 396 $\mu g \cdot L^{-1}$ in summer slightly higher than the 200 - 300 $\mu g \cdot L^{-1}$ prefertilization levels from Saqvaqjuac lakes (Welch et al. 1989). Silica measured in the Lac de Gras lakes was slightly higher in Fish2 Lake than in Fish1 Lake (Table 3) and the range of 79 - 272 $\mu g \cdot L^{-1}$ exceeded the 100 $\mu g \cdot L^{-1}$ levels typical of prefertilization open water values in Saqvaqjuac lakes (Welch et al. 1989). Suspended C was measured all three years and ranged from 100 - 1180 $\mu g \cdot L^{-1}$. Suspended Fe ranged from 18 - 134 $\mu g \cdot L^{-1}$ over the three years. The concentration of Chl (Fig. 7) was measured from mid-July until the end of August in 1995, and from late June until the end of August in 1996. Chl remained relatively uniform for both lakes except following ice-off until mid-July 1996 in Fish2 Lake. This early period was not sampled in 1995. Peak production was found to occur anytime from before ice-out to August at Saqvaqjuac, and varied greatly depending on snow and slush ice cover during a particular year (Welch et al. 1989). Average concentration in Fish1 Lake in 1995 was 1.27 $\mu g \cdot L^{-1}$ (range 0.78 - 1.60) and in 1996 was 1.60 $\mu g \cdot L^{-1}$ (range 0.62 - 11.26). In 1995, Fish2 Lake concentration averaged 1.17 $\mu g \cdot L^{-1}$ (range 0.78 - 1.60). In 1996, from 25 June to 11 July, the concentration averaged 10.93 $\mu g \cdot L^{-1}$ (range 5.80 - 22.10) and from 15 July to 27 August had dropped to 1.14 $\mu g \cdot L^{-1}$ (range 0.54 - 1.66).

ZOOPLANKTON

The zooplankton communities were studied in 1994 and 1995 (Fig. 8 and 9). Sampling began in late July in 1994 and in early July 1995. In both lakes the Cyclopoida nauplii were the most important components of the communities based on

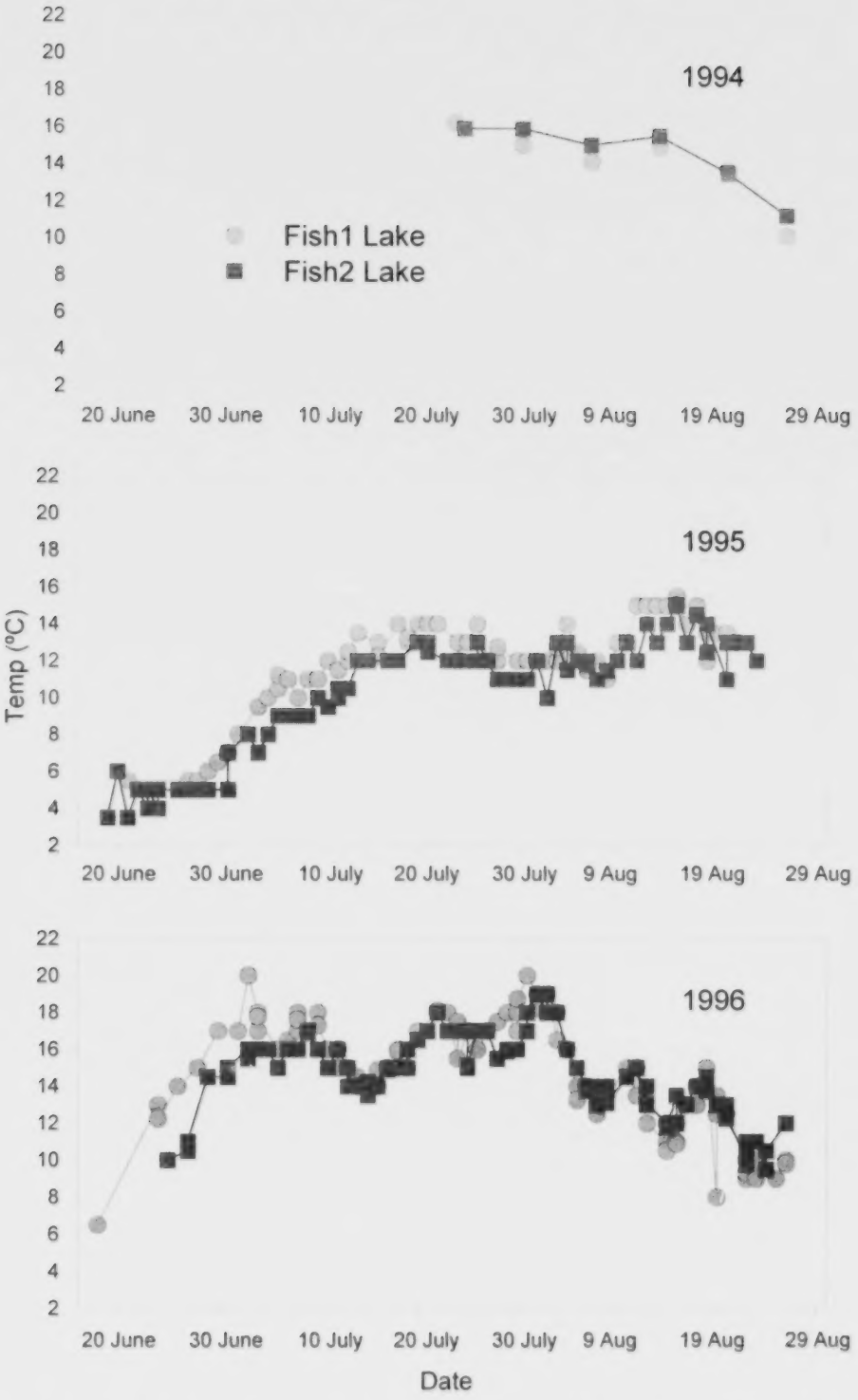

Fig. 4. Surface water temperatures for Fish1 and Fish2 lakes, 1994 - 1996.

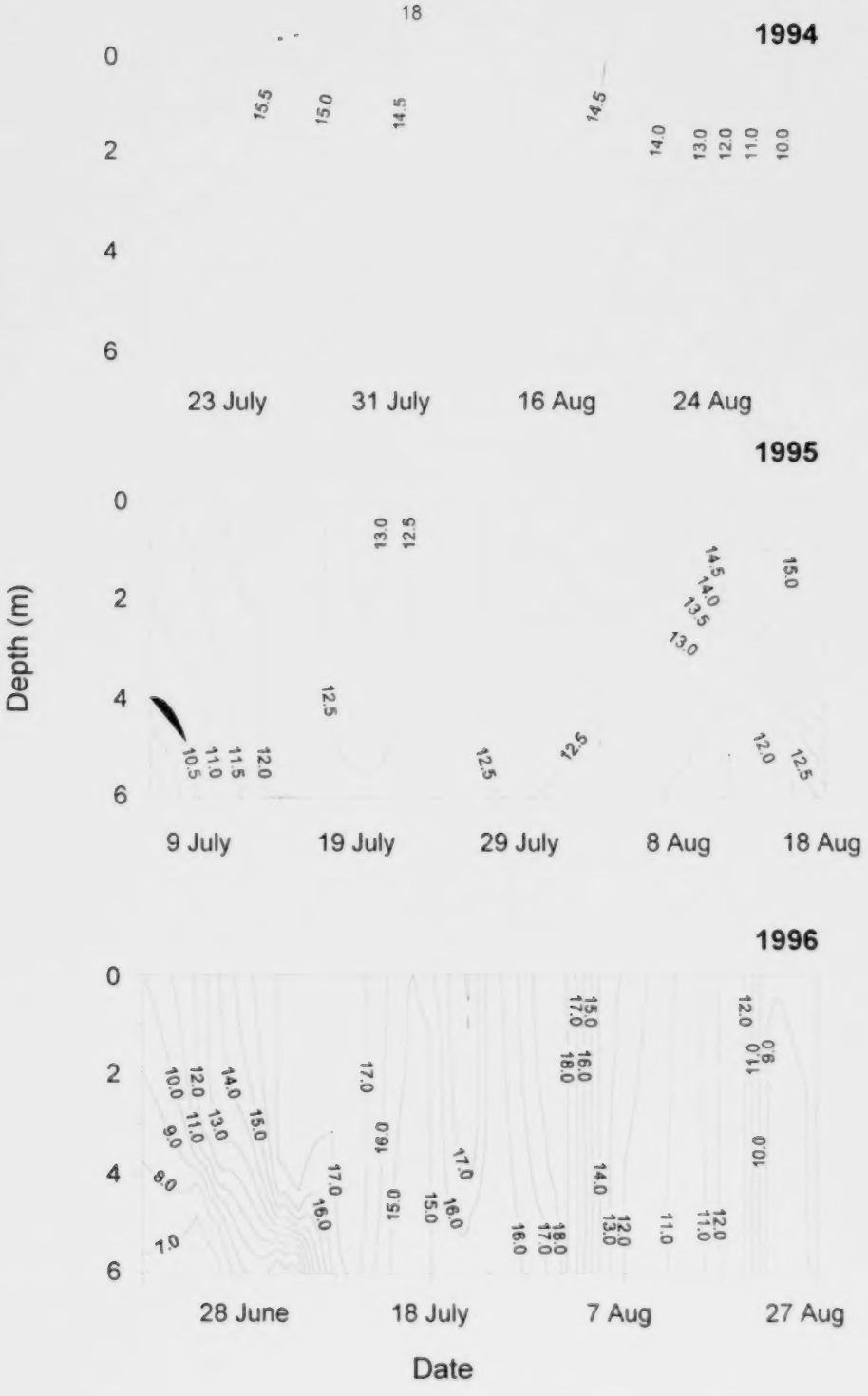

Fig. 5. Temperature isopleths for Fish1 Lake, 1994 - 1996.

Fig. 6. Temperature isopleths for Fish2 Lake, 1994 - 1996.

Table 3. Summary of mean concentrations (± SD) of total dissolved N, total dissolved P, soluble reactive Si, suspended N, suspended C, chlorophyll-*a*, and suspended Fe from summer sampling in Fish1 and Fish2 lakes.

Lake	Year	Depth (m)	TDN ($\mu g \cdot L^{-1}$)	TDP ($\mu g \cdot L^{-1}$)	SRSI ($\mu g \cdot L^{-1}$)	SUSP N ($\mu g \cdot L^{-1}$)	SUSP P ($\mu g \cdot L^{-1}$)	SUSP C ($\mu g \cdot L^{-1}$)	CHL-*a* ($\mu g \cdot L^{-1}$)	SUSP FE ($\mu g \cdot L^{-1}$)
Fish1	1994	1	217 ± 83	1 ± 1	92 ± 13	70 ± 7	11 ± 1	553 ± 70		113 ± 22
	1995	1	140 ± 46	5 ± 1	147 ± 7	63 ± 18	6 ± 1	536 ± 95	1.19 ± 0.19	49 ± 5
		4	181 ± 75	5 ± 1	147 ± 9	65 ± 13	7 ± 1	527 ± 91	1.34 ± 0.19	50 ± 4
	1996	1	128 ± 23	4 ± 0	179 ± 18	62 ± 12	6 ± 1	521 ± 90	1.79 ± 2.18	66 ± 16
		4	125 ± 27	4 ± 1	177 ± 17	66 ± 14	6 ± 1	568 ± 187	1.40 ± 0.49	67 ± 16
Fish2	1994	1	187 ± 48	< 1	165 ± 34	49 ± 5	7 ± 1	427 ± 50		45 ± 3
	1995	1	120 ± 46	4 ± 1	179 ± 7	50 ± 20	4 ± 1	427 ± 113	1.11 ± 0.39	21 ± 3
		10	110 ± 33	4 ± 1		42 ± 2	4 ± 1	376 ± 25	1.24 ± 0.30	22 ± 3
	1996	1	115 ± 88	3 ± 1	229 ± 16	41 ± 6	4 ± 1	362 ± 98	2.54 ± 2.58	25 ± 5
		4	90 ± 33	3 ± 1	227 ± 19	44 ± 7	4 ± 1	387 ± 109	4.85 ± 6.86	25 ± 3

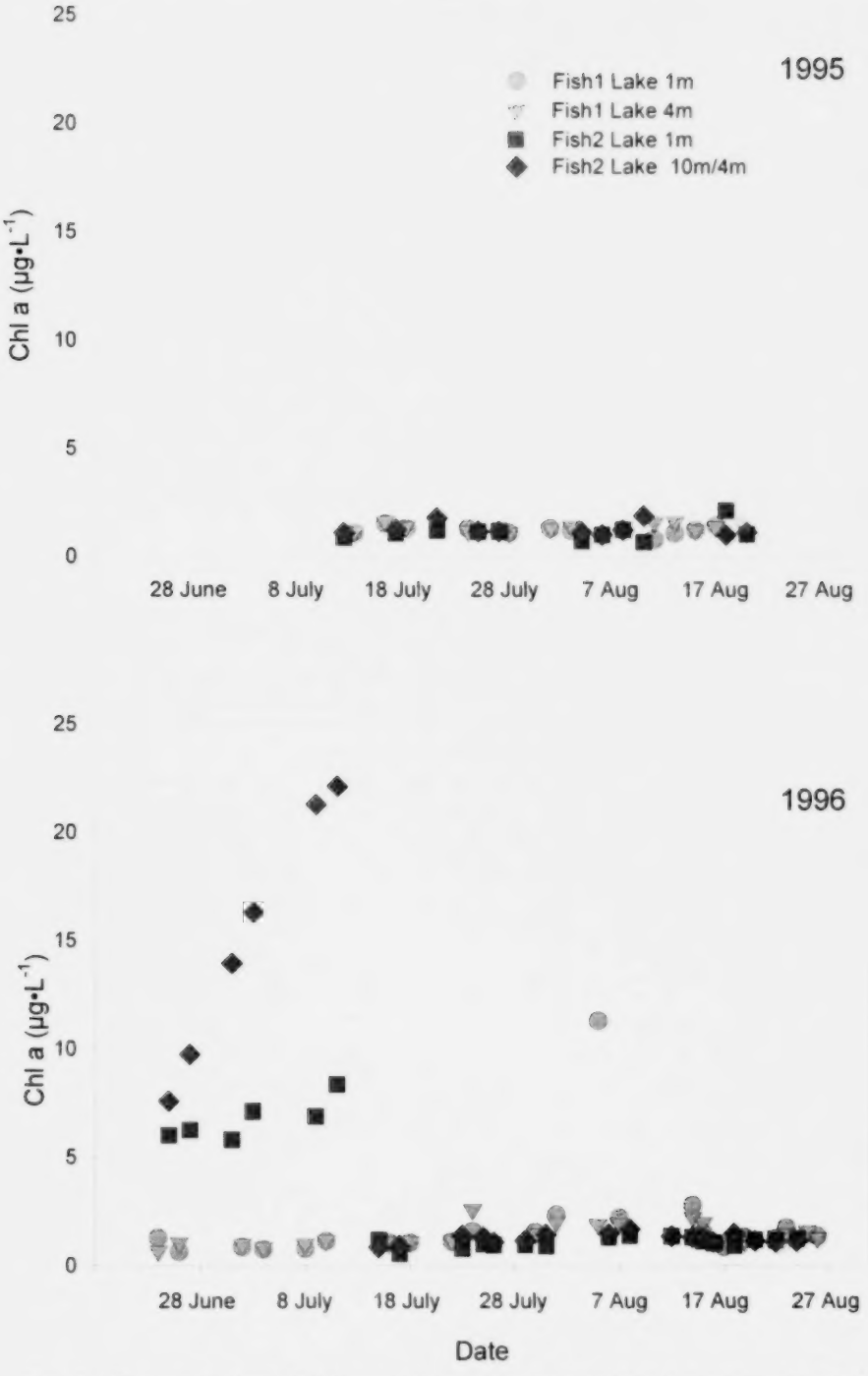

Fig. 7. Concentration of chlorophyll-*a* measured during the summers of 1995 and 1996 for Fish1 and Fish2 lakes. Samples were taken from two depths: 1 m and 4 m both years in Fish1 Lake, and 1 m and 10 m in 1995 and 1 m and 4 m in 1996 in Fish2 Lake.

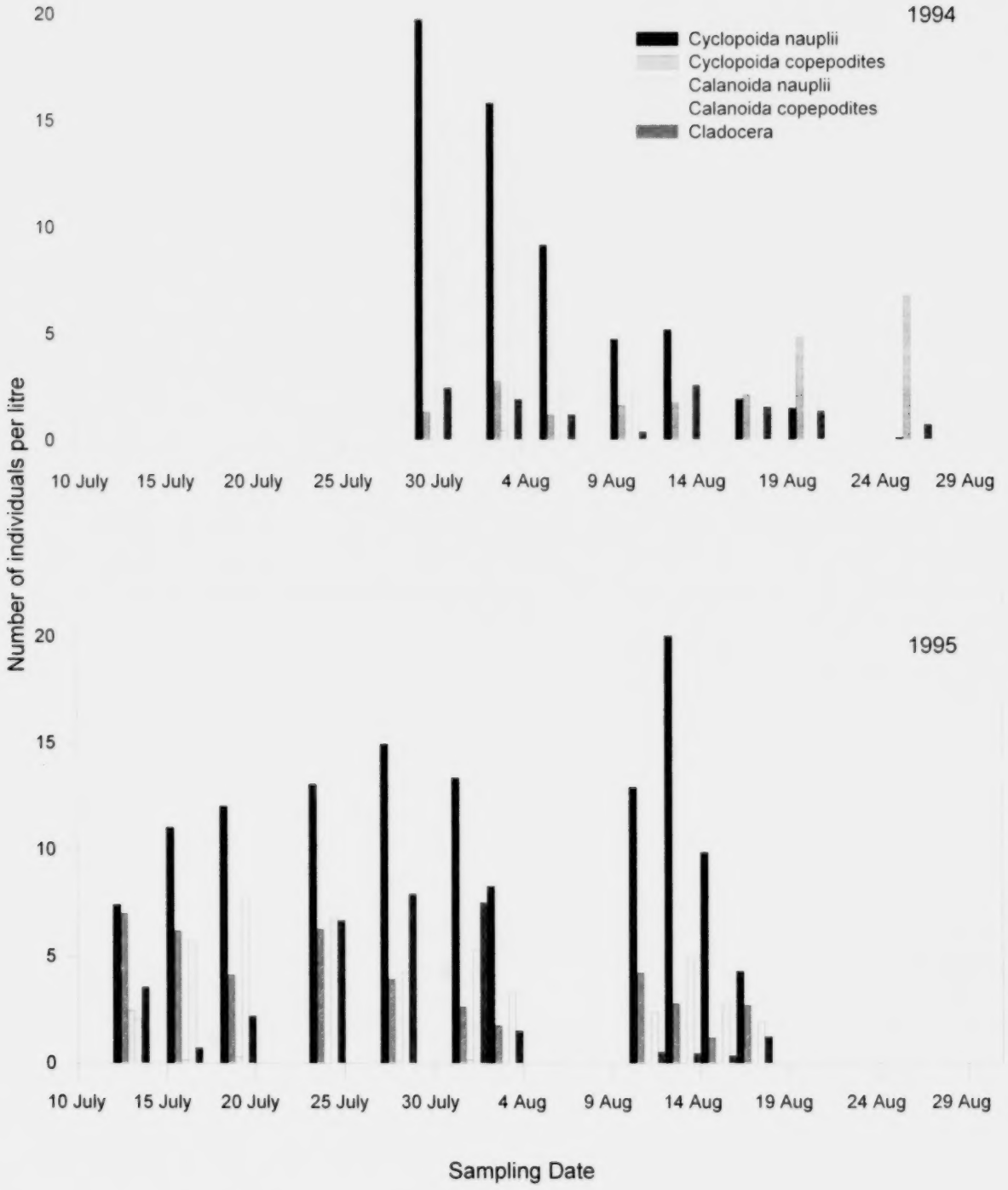

Fig. 8. Abundance of major zooplankton groups in Fish1 Lake, 1994 and 1995.

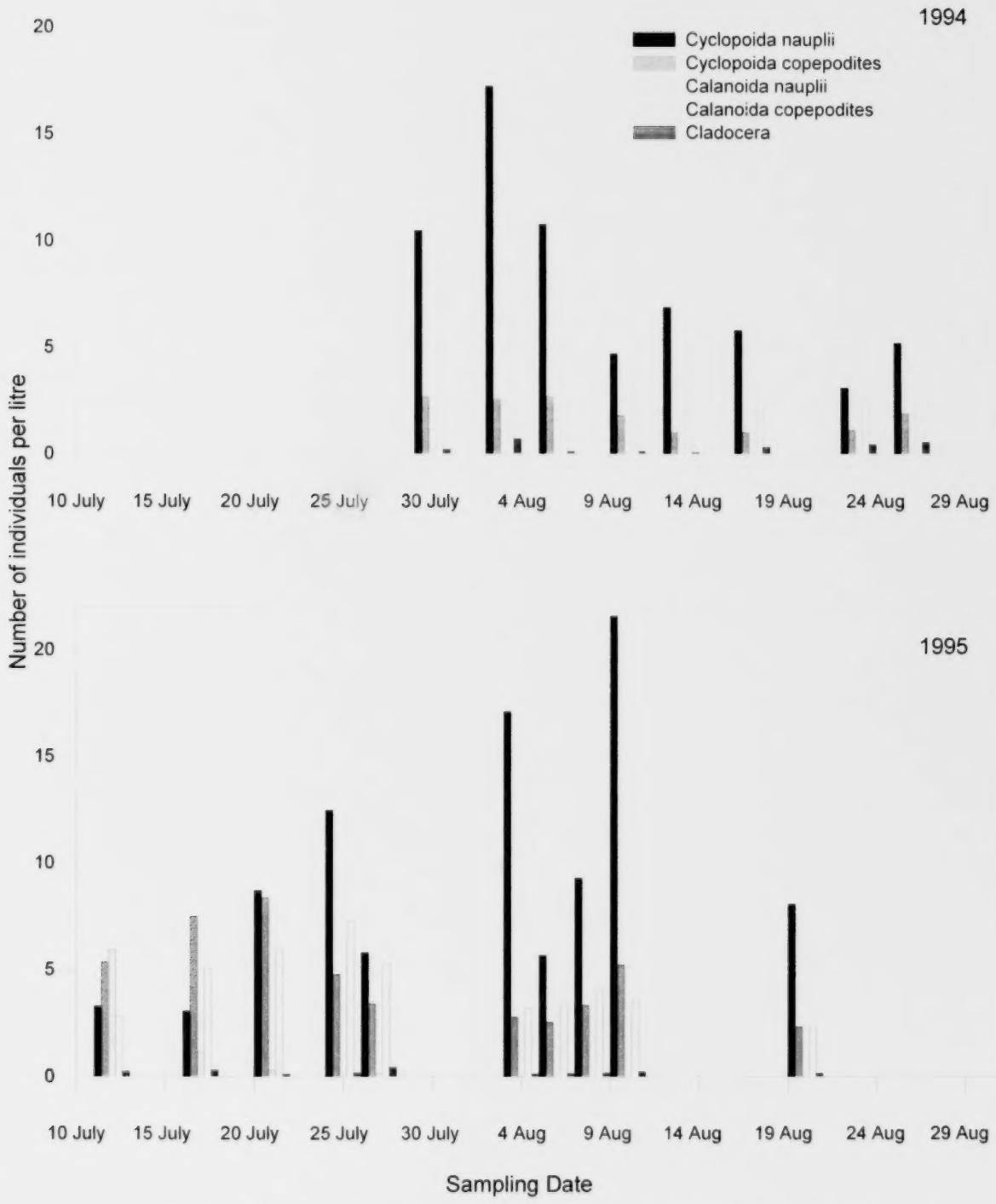

Fig. 9. Abundance of major zooplankton groups in Fish2 Lake, 1994 and 1995.

number of individuals per litre over both years followed by Calanoida copepodites (Fig. 8 and 9). Cyclopoida copepodites made up a larger proportion of the collection in Fish1 Lake increasing as the number of Cyclopoida nauplii decreased in August 1994. In Fish2 Lake, the nauplii didn't decrease to the same extent, nor did the number of copepodites increase as much over the same period. In Fish2 Lake in 1995, the Cyclopoida copepodites were important in July, outnumbering the nauplii at the start of the month. Calanoida nauplii were most numerous in the first samples of July 1995 in both lakes, but were a minor component of the communities for the remainder of the time. Cladocera were much more prevalent in Fish1 Lake than in Fish2 Lake in both years, and made up a higher proportion of the catch in 1995.

Detailed species composition of the communities varied between lakes and between years (Tables 4, 5, 6, and 7). In Fish1 Lake, three species of cyclopoid copepods were identified (Tables 4 and 5). *Cyclops scutifer* was collected consistently in both 1994 and 1995. *Macrocyclops albidus*, a fairly large cyclopoid copepod, was not collected at all in 1994 but was collected throughout 1995. *Eucyclops speratus* was collected only in the last 1994 sample and in half of the 1995 samples. In Fish2 Lake, only two species of cyclopoid copepods were identified (Table 6 and 7). *C. scutifer* was the dominant species in both years. In 1994, only immature individuals were collected, likely because sampling didn't begin until late July, by which time the adult numbers had tapered off as they had in 1995. *C. capillatus*, the other species collected, was only found in two samples in 1995.

The calanoid copepod community of Fish1 Lake was made up of six species (Tables 4 and 5). *Epischura lacustris* were collected in both years. *E. nevadensis* was only identified in the July 30, 1994 sample. *Diaptomus pribilofensis* was collected in both years, although it was most important in the August samples. Immature *D. sicilis* were collected in all samples while adults were only identified in the late August 1994 samples. *Heterocope septentrionales* adult females were identified in all samples of 1994, while no immature copepodites were identified for this species. In 1995, immature copepodites were found in all samples, and adult females were collected from 24 July on. *D. ashlandi* were only identified in the last sample of 1994. The calanoid copepod community of Fish2 Lake was made up of three species: *D. pribilofensis*, *D. sicilis*, and *H. septentrionales*. In 1994, the first two species were collected in each sample (*D. sicilis* were all immature), and the latter species was collected in just over half the samples. In 1995, *D. sicilis* were collected in all samples. Immature individuals were identified in all samples and adults in the July and early August samples. Immature *H. septentrionales* were identified in the July samples with adults identified in the last July sample and all the August samples.

The Cladocera community was made up of four species in Fish1 Lake (Table 4 and 5), and was dominated by *Holopedium gibberum*, which were collected in all samples. *Daphnia middendorffiana* were collected throughout 1995, but were only found in three of the 1994 samples. The other two species were collected sporadically in both years. In Fish2 Lake, there were five species of Cladocera collected (Table 6 and 7), none of which attained the density found in Fish1 Lake. *H. gibberum* were sporadic in 1994 and collected in all but one sample of 1995. *Eubosmina longispina* were identified in six of eight 1994 samples, and three of ten 1995 samples. *D. middendorffiana* were found in most samples in both 1994 and 1995. *D. longiremis* were found in only three samples from 1994, but in most samples from 1995. *Chydorus sphaericus* was only identified in one 1994 sample.

Table 4. Zooplankton community (number of individuals per L) from Fish1 Lake in 1994. Zooplankton are identified as immature (I), male (M), female (F), female with eggs (F+), nauplii (N), or adult (A).

Species	Sex	Jul 30	Aug 3	Aug 6	Aug 10	Aug 13	Aug 17	Aug 20	Aug 26
Macrocyclops albidus	I	-	-	-	-	-	-	-	-
Eucyclops speratus	F	-	-	-	-	-	-	-	-
Eucyclops speratus	M	-	-	-	-	-	-	-	0.072
Eucyclops speratus	I	-	-	-	-	-	-	-	-
Cyclops capillatus	F	-	-	-	-	-	-	-	-
Cyclops capillatus	M	-	-	-	-	-	-	-	-
Cyclops scutifer	F	-	-	-	-	-	-	-	-
Cyclops scutifer	F+	-	-	-	0.072	-	-	0.072	-
Cyclops scutifer	M	-	-	-	-	-	-	-	0.001
Cyclops scutifer	I	1.303	2.750	1.158	1.520	1.737	2.099	4.705	6.659
Cyclopoida	N/small	19.687	15.779	9.120	4.705	5.139	1.882	1.448	0.072
Epischura lacustris	A	0.006	0.001	-	-	-	0.004	0.116	0.004
Epischura lacustris	I	-	-	-	-	0.072	-	0.058	-
Epischura nevadensis	A	0.001	-	-	-	-	-	-	-
Diaptomus pribilofensis	F	0.145	1.013	1.158	0.434	0.362	-	0.145	0.072
Diaptomus pribilofensis	F+	-	0.003	-	-	0.049	-	0.116	0.012
Diaptomus pribilofensis	M	0.145	0.434	0.290	0.507	0.362	0.145	0.145	-
Diaptomus pribilofensis	I	4.343	4.777	0.579	0.0362	0.072	-	0.072	-
Diaptomus sicilis	F	-	-	-	-	-	0.072	0.072	-
Diaptomus sicilis	M	-	-	-	-	-	-	-	0.145
Diaptomus sicilis	I	3.329	3.474	1.158	1.592	1.375	0.941	1.737	1.375
Heterocope septentrionales	F	0.020	0.025	0.045	0.006	0.016	0.010	0.014	0.012
Heterocope septentrionales	F+	0.006	0.009	0.022	-	0.003	0.001	0.004	0.003
Heterocope septentrionales	I	-	-	-	-	-	-	-	-
Diaptomus ashlandi	F	-	-	-	-	-	-	-	0.002
Diaptomus ashlandi	M	-	-	-	-	-	-	-	0.002
Calanoida	N/small	-	0.434	-	-	-	-	-	-
Calanoida	N/large	-	-	-	-	-	-	-	-
Eubosmina longispina	F	-	0.145	-	0.058	0.058	-	0.001	-
Eubosmina longispina	F+	-	-	-	-	-	-	0.003	-
Eubosmina longispina	M	-	-	-	-	-	-	-	-
Eubosmina longispina	I	-	0.145	-	-	-	-	0.058	-
Daphnia middendorffiana	F	-	-	0.003	-	-	-	-	-
Daphnia middendorffiana	F+	0.004	-	-	-	-	-	-	-
Daphnia middendorffiana	M	-	-	-	-	-	-	-	0.001
Daphnia middendorffiana	I	0.001	-	-	-	-	-	0.001	-
Chydorus sphaericus	F	-	-	-	-	-	-	-	-
Daphnia longiremis	F	-	-	-	-	-	-	-	-
Daphnia longiremis	F+	-	-	-	-	-	-	-	-
Daphnia longiremis	I	-	-	-	-	-	-	-	-
Holopedium gibberum		2.432	1.592	1.158	0.290	2.490	1.506	1.274	0.695

Table 5. Zooplankton community (number of individuals per L) from Fish1 Lake in 1995. Zooplankton are identified as immature (I), male (M), female (F), female with eggs (F+), nauplii (N), or adult (A).

Species	Sex	Jul 13	Jul 16	Jul 19	Jul 24	Jul 28	Aug 1	Aug 3	Aug 11	Aug 13	Aug 15	Aug 17
Macrocyclops albidus	I	4.777	3.040	0.434	2.750	1.882	1.303	0.290	1.013	1.158	0.290	1.013
Eucyclops speratus	F	-	-	-	-	-	-	-	0.145	-	0.145	-
Eucyclops speratus	M	-	-	-	-	-	-	-	-	0.145	-	-
Eucyclops speratus	I	0.145	0.290	-	0.290	-	-	-	-	-	-	-
Cyclops capillatus	F	-	-	-	-	-	-	-	-	-	-	-
Cyclops capillatus	M	-	-	-	-	-	-	-	-	-	-	-
Cyclops scutifer	F	0.290	-	0.434	0.174	0.145	-	0.145	0.145	-	-	-
Cyclops scutifer	F+	0.174	0.232	0.058	-	-	0.145	-	-	-	-	-
Cyclops scutifer	M	-	-	-	-	-	-	-	0.145	-	-	-
Cyclops scutifer	I	1.592	2.606	3.1857.	3.040	1.882	1.158	1.303	2.750	1.448	0.724	1.665
Cyclopoida	N/small	7.383	11.002	12.015	13.028	14.910	13.318	8.251	12.884	19.977	9.844	4.270
Epischura lacustris	A	-	-	-	-	-	-	0.029	-	0.290	-	-
Epischura lacustris	I	-	0.058	0.058	0.579	0.290	0.290	0.029	-	-	-	-
Epischura nevadensis	A	-	-	-	-	-	-	-	-	-	-	-
Diaptomus pribilofensis	F	-	-	-	-	-	-	-	0.145	0.869	1.013	0.869
Diaptomus pribilofensis	F+	-	-	-	-	-	-	-	-	-	-	-
Diaptomus pribilofensis	M	-	-	-	-	-	-	-	0.145	0.145	0.579	0.362
Diaptomus pribilofensis	I	-	-	0.290	4.343	2.461	2.461	2.316	0.434	0.145	0.145	0.145
Diaptomus sicilis	F	-	-	-	-	-	-	-	-	-	-	-
Diaptomus sicilis	M	-	-	-	-	-	-	-	-	-	-	-
Diaptomus sicilis	I	1.737	5.646	7.238	1.737	1.013	2.171	0.869	1.592	3.040	1.013	0.434
Heterocope septentrionales	F	-	-	-	-	0.232	0.145	0.052	0.039	0.088	0.038	0.012
Heterocope septentrionales	F+	-	-	-	0.058	0.058	0.174	0.022	0.009	0.019	0.019	0.014
Heterocope septentrionales	I	0.347	0.058	0.116	0.058	0.174	0.029	0.004	0.029	0.290	-	0.072
Diaptomus ashlandi	F	-	-	-	-	-	-	-	-	-	-	-
Diaptomus ashlandi	M	-	-	-	-	-	-	-	-	-	-	-
Calanoida	N/small	2.461	0.145	0.290	-	-	0.145	-	-	-	-	-
Calanoida	N/large							-	-	-	-	-
Eubosmina longispina	F	-	0.145	-	-	-	0.029	-	-	-	-	-
Eubosmina longispina	F+	-	-	0.001	-	0.001	0.029	-	-	-	-	-
Eubosmina longispina	M	-	-	-	-	-	-	-	-	-	-	-
Eubosmina longispina	I	-	0.145	-	-	0.058	0.029	-	-	-	-	-
Daphnia middendorffiana	F	-	-	-	0.058	0.006	0.001	0.029	-	-	-	-
Daphnia middendorffiana	F+	0.001	-	-	0.174	0.004	0.010	-	-	-	-	-
Daphnia middendorffiana	M	-	-	-	-	-	-	-	-	-	-	-
Daphnia middendorffiana	I	0.116	-	0.145	0.174	0.001	0.145	0.001	-	0.145	-	-
Chydorus sphaericus	F	-	-	-	-	-	-	-	-	-	-	-
Daphnia longiremis	F	-	-	-	-	0.003	-	-	-	-	-	-
Daphnia longiremis	F+	-	-	-	-	0.001	-	-	-	-	-	-
Daphnia longiremis	I	-	-	-	-	0.434	-	-	0.145	-	0.145	-
Holopedium gibberum		3.416	0.405	2.027	6.254	7.383	7.238	1.448	0.347	0.290	0.174	1.216

Table 6. Zooplankton community (number of individuals per L) from Fish2 Lake in 1994. Zooplankton are identified as immature (I), male (M), female (F), female with eggs (F+), nauplii (N), or adult (A).

Species	Sex	Jul 30	Aug 3	Aug 6	Aug 10	Aug 13	Aug 17	Aug 23	Aug 26
Macrocyclops albidus	I	-	-	-	-	-	-	-	-
Eucyclops speratus	F	-	-	-	-	-	-	-	-
Eucyclops speratus	M	-	-	-	-	-	-	-	-
Eucyclops speratus	I	-	-	-	-	-	-	-	-
Cyclops capillatus	F	-	-	-	-	-	-	-	-
Cyclops capillatus	M	-	-	-	-	-	-	-	-
Cyclops scutifer	F	-	-	-	-	-	-	0.100	-
Cyclops scutifer	F+	-	-	-	-	-	-	-	-
Cyclops scutifer	M	-	-	-	-	-	-	-	-
Cyclops scutifer	I	2.687	2.588	2.687	1.791	0.995	0.995	0.995	1.891
Cyclopoida	N/small	10.450	17.217	10.748	4.678	6.867	5.772	3.085	5.175
Epischura lacustris	A	-	-	-	-	-	-	-	-
Epischura lacustris	I	-	-	-	-	-	-	-	-
Epischura nevadensis	A	-	-	-	-	-	-	-	-
Diaptomus pribilofensis	F	-	-	0.100	-	0.697	0.796	1.393	0.697
Diaptomus pribilofensis	F+	-	-	-	-	-	-	0.326	0.605
Diaptomus pribilofensis	M	-	-	-	0.299	0.597	0.796	1.393	0.398
Diaptomus pribilofensis	I	3.583	4.080	3.185	2.488	1.493	0.199	0.100	0.100
Diaptomus sicilis	F	0.100	-	-	-	-	-	-	-
Diaptomus sicilis	M	-	-	0.100	-	-	-	-	-
Diaptomus sicilis	I	1.393	2.189	1.990	2.787	2.389	0.995	0.896	1.791
Heterocope septentrionales	F	0.299	-	-	0.100	0.014	-	0.016	0.056
Heterocope septentrionales	F+	-	-	-	-	-	-	-	0.008
Heterocope septentrionales	I	-	-	-	-	-	-	-	-
Diaptomus ashlandi	F	-	-	-	-	-	-	-	-
Diaptomus ashlandi	M	-	-	-	-	-	-	-	-
Calanoida	N/small	-	-	0.100	-	-	-	-	-
Calanoida	N/large	-	-	-	-	-	-	-	0.100
Eubosmina longispina	F	-	0.199	-	-	0.020	0.100	0.100	0.199
Eubosmina longispina	F+	-	0.100	-	-	0.006	-	0.040	-
Eubosmina longispina	M	-	0.100	-	-	-	-	-	-
Eubosmina longispina	I	0.199	-	-	-	0.004	-	-	0.100
Daphnia middendorffiana	F	-	0.100	-	-	0.010	-	-	-
Daphnia middendorffiana	F+	-	-	0.100	-	0.002	-	0.064	0.127
Daphnia middendorffiana	M	-	-	-	-	0.002	-	-	-
Daphnia middendorffiana	I	-	-	-	0.100	0.022	0.199	0.100	0.100
Chydorus sphaericus	F	-	-	-	-	-	-	0.100	-
Daphnia longiremis	F	-	-	-	-	-	-	-	-
Daphnia longiremis	F+	-	0.100	-	-	-	-	-	0.008
Daphnia longiremis	I	-	-	-	-	0.002	-	-	-
Holopedium gibberum		-	0.100	-	-	0.002	-	0.016	-

Table 7. Zooplankton community (number of individuals per L) from Fish2 Lake in 1995. Zooplankton are identified as immature (I), male (M), female (F), female with eggs (F+), nauplii (N), or adult (A).

Species	Sex	Jul 12	Jul 17	Jul 21	Jul 25	Jul 27	Aug 4	Aug 6	Aug 8	Aug 10	Aug 20
Macrocyclops albidus	I	-	-	-	-	-	-	-	-	-	-
Eucyclops speratus	F	-	-	-	-	-	-	-	-	-	-
Eucyclops speratus	M	-	-	-	-	-	-	-	-	-	-
Eucyclops speratus	I	-	-	-	-	-	-	-	-	-	-
Cyclops capillatus	F	0.072	-	-	-	-	-	-	-	-	-
Cyclops capillatus	M	-	-	-	-	0.145	-	-	-	-	-
Cyclops scutifer	F	0.579	0.651	1.448	0.434	0.290	0.145	0.072	0.145	-	-
Cyclops scutifer	F+	0.217	0.543	0.398	0.145	0.054	0.018	-	-	-	-
Cyclops scutifer	M	0.290	0.145	0.145	-	-	-	-	-	-	-
Cyclops scutifer	I	4.198	6.152	6.369	4.198	2.895	2.606	2.461	3.185	5.211	2.316
Cyclopoida	N/small	3.257	3.040	8.686	12.449	5.790	17.082	5.648	9.265	21.569	8.034
Epischura lacustris	A	-	-	-	-	-	-	-	-	-	-
Epischura lacustris	I	-	-	-	-	-	-	-	-	-	-
Epischura nevadensis	A	-	-	-	-		-	-	-	-	-
Diaptomus pribilofensis	F	-	-	-	-	-	-	-	-	0.018	0.651
Diaptomus pribilofensis	F+	-	-	-	-	-	-	-	-	-	-
Diaptomus pribilofensis	M	-	-	-	-	-	-	-	-	-	0.651
Diaptomus pribilofensis	I	-	-	-	-	0.579	1.158	1.665	3.185	2.027	0.217
Diaptomus sicilis	F	-	-	0.145	-	-	-	-	-	0.018	-
Diaptomus sicilis	M	-	0.072	0.434	0.145	-	0.145	0.145	0.145	0.018	-
Diaptomus sicilis	I	2.750	4.849	5.356	7.093	4.632	1.882	1.520	0.724	1.448	0.796
Heterocope septentrionales	F	-	-	-	-	-	0.036	-	-	0.018	0.018
Heterocope septentrionales	F+	-	-	-	-	0.054	-	0.036	0.018	0.018	-
Heterocope septentrionales	I	0.054	0.145	0.036	0.018	0.018	-	-	-	0.018	-
Diaptomus ashlandi	F	-	-	-	-	-	-	-	-	-	-
Diaptomus ashlandi	M	-	-	-	-	-	-	-	-	-	-
Calanoida	N/small	5.863	1.158	0.290	-	0.145	-	-	-	-	-
Calanoida	N/large	0.072					-	-	-	-	-
Eubosmina longispina	F	-	-	-	-	0.018	-	0.018	-	-	0.018
Eubosmina longispina	F+	-	-	-	-	-	-	-	-	-	-
Eubosmina longispina	M	-	-	-	-	-	-	-	-	-	-
Eubosmina longispina	I	-	-	-	-	-	-	-	-	-	-
Daphnia middendorffiana	F	-	-	-	-	0.145	-	-	0.018	-	-
Daphnia middendorffiana	F+	-	-	-	-	-	0.018	-	0.018	-	0.018
Daphnia middendorffiana	M	-	-	-	-	-	-	-	-	-	-
Daphnia middendorffiana	I	-	0.018	-	0.036	0.018	0.018	0.018	0.072	0.018	0.018
Chydorus sphaericus	F	-	-	-	-	-	-	-	-	-	-
Daphnia longiremis	F	0.072	0.018	-	0.018	0.036	0.018	-	0.036	0.072	0.072
Daphnia longiremis	F+	-	-	-	-	-	-	-	-	-	-
Daphnia longiremis	I	-	0.018	-	-	0.018	-	-	-	-	-
Holopedium gibberum		0.145	0.217	0.090	0.109	0.181	0.054	0.090	0.018	0.127	-

Based on bulk zooplankton samples in 1994, 1995, and 1996, Fish1 Lake produced higher biomass of zooplankton per litre than did Fish2 Lake (Fig. 10). In 1994 the mean sample weight per litre (± 1 standard deviation) was 0.0929 $mg \cdot L^{-1}$ (± 0.0313) in Fish1 Lake and 0.0819 $mg \cdot L^{-1}$ (±0.0219) in Fish2 Lake. In 1995, the mean sample weight per liter was 0.1115 $mg \cdot L^{-1}$ (±0.0369) in Fish1 Lake and 0.0445 $mg \cdot L^{-1}$ (±0.0162) in Fish2 Lake. In 1996, the mean sample weight per liter was 0.0857 $mg \cdot L^{-1}$ (±0.0555) in Fish1 Lake and 0.0491 $mg \cdot L^{-1}$ (±0.0297) in Fish2 Lake.

INSECT EMERGENCE

Emergence was measured in 1995 and 1996 (Table 8), and for both years the majority of the emerging insects collected by the traps were Diptera. Other insects collected occasionally included Trichoptera, Ephemeroptera, Plecoptera, Coleoptera, and Hemiptera. Emerging Diptera were collected from all depths. In 1995, sampling for emergence did not begin until 7 July in Fish1 Lake, and 13 July in Fish2 Lake (Fig. 11). In 1996, emergence traps were placed in the lake as soon as possible prior to the ice being completely gone from the lakes: 22 June in Fish1 Lake and 15 June in Fish2 Lake (Fig. 12). The earlier sampling dates in 1996 resulted in the collection of the earliest peak of Diptera emergence missed in the 1995 sampling. The pattern of emergence differed between the two lakes. In Fish2 Lake, larger proportions of the emergence occurred immediately after ice out. Over the remainder of the sampling period, the emergence was steady but at much lower levels. In Fish1 Lake, a lower proportion of the emergence occurred in the early spring peak (Fig. 12), and as a result, the estimates of total emergence (Table 8) are much closer between the two years for Fish1 Lake than Fish2 Lake. For comparison purposes, the 1996 data were better estimates of the total Diptera emergence. Size of emerging Diptera varied throughout the season and between lakes, which is likely a reflection of the species make-up. The emergence from Fish1 and Fish2 lakes was compared to that reported from the Saqvaqjuac lakes (Table 8) (see also Welch et al. 1988). The number of Diptera emerging in the two Lac de Gras area lakes was higher than that found in all of the Saqvaqjuac lakes in terms of number per m^2. It was as much as 3 x higher than the pre-fertilization emergence from P&N Lake, and slightly higher when compared to the post-fertilization years from P&N Lake (1979-1981). The average size of the emerging adults was smaller, and as a result, the overall emerging biomass from the Lac de Gras lakes was 2.8 x that found in the pre-fertilization P&N Lake, but 0.85 x that found in the post-fertilization years from P&N Lake. The emerging biomass was also 0.85 x the average emergence from Spring Lake, slightly higher than that of Jade Lake, and 1.3 x that from Far Lake.

FISH POPULATIONS

The two study lakes (Fish1 and Fish2 lakes) contained lake trout, round whitefish, arctic grayling, and burbot. Slimy sculpin *(Cottus cognatus)* were also collected in both lakes. One lake chub (*Couesius plumbeus*) and several shiners, likely emerald or spottail shiners (*Notropis sp*), were collected from Fish1 Lake. Other lakes in the area were also found to contain longnose sucker (*Catostomus catostomus*) and lake whitefish (*Coregonus clupeaformis*) (Rescan 1994a). A total of 144 fish (not including burbot) were removed from several lakes in the area between 1994 and 1996 as part of this study, and were principally used for contaminant analysis. No attempt was made to sample or estimate abundance or biomass of Cyprinidae or sculpin.

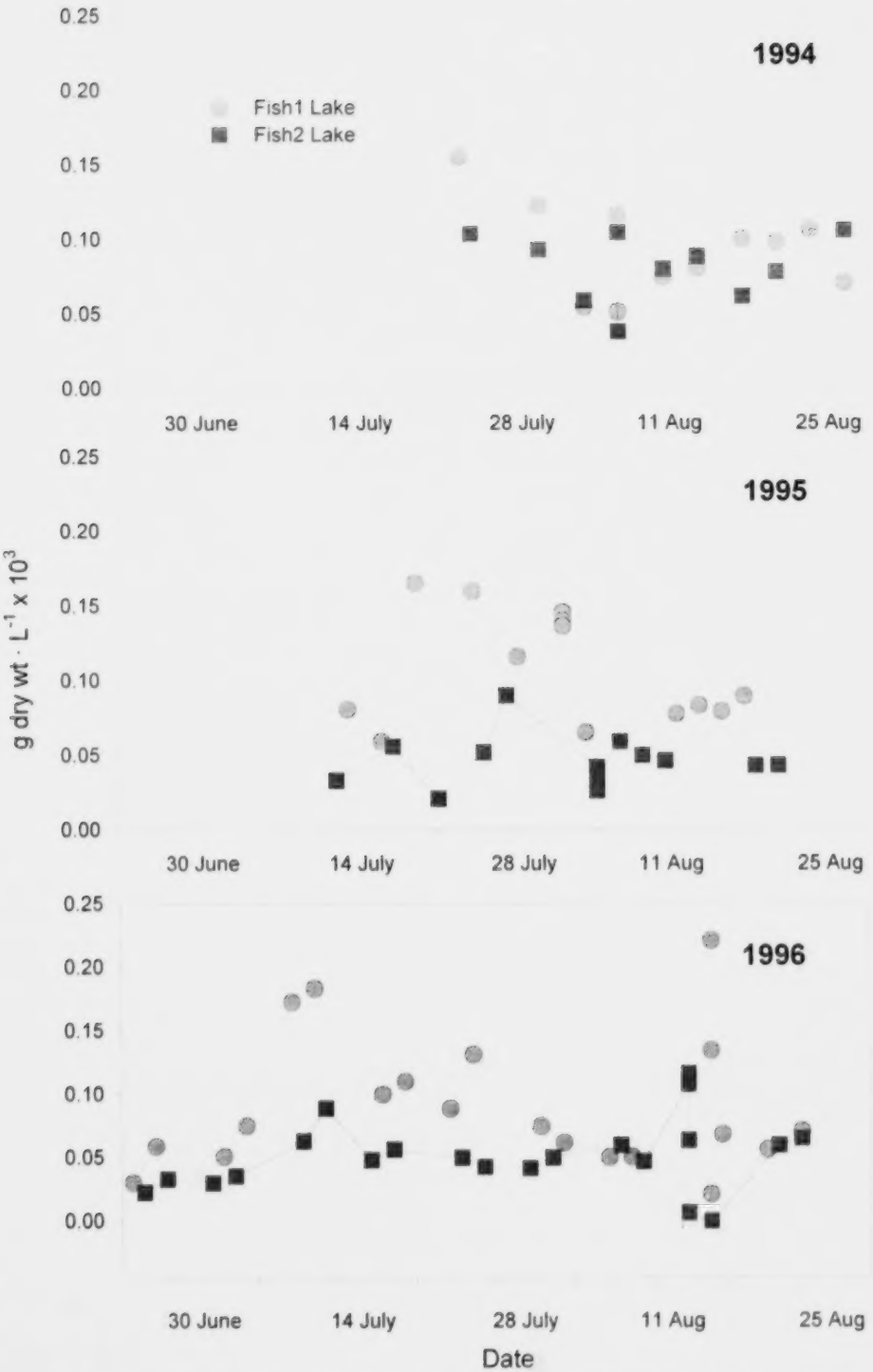

Fig. 10. Zooplankton biomass for Fish1 and Fish2 lakes, 1994 - 1996.

Table 8. Summary of Diptera emergence measured in number of individuals and dry weight for Fish1 and Fish2 lakes in 1995 and 1996. Emergence from four Saqvaqjuac lakes (Welch et al. 1988) is presented for comparison.

Lake	Area (ha)	Max Depth (m)	Year	Sampling Season	Total No. Emerging ($\times 10^6$)	Number Emerging ($no.\cdot m^{-2}\cdot y^{-1}$)	Mean dry Weight per adult (mg)	Total weight (Kg)	Emerging Biomass ($g\cdot m^{-2}\cdot y^{-1}$)
Fish1	45.2	6.0	1995	7 July – 21 August	928.9	2055	0.233	118.2	0.262
			1996	22 June – 26 August	1060.0	2345	0.151	160.7	0.356
Fish2	68.0	18.0	1995	13 July – 20 August	598.4	880	0.268	80.7	0.119
			1996	15 June – 26 August	1468.1	2159	0.121	241.5	0.355
P&N	7.1	10.2	1977-1978		52.8	743	0.170		0.126
			1979-1981		136.4	1921	0.218		0.421
Far	3.7	8.9	1977-1981		41.0	1108	0.223		0.269
Spring	6.9	7.1	1979-1981		112.6	1632	0.268		0.419
Jade	3.6	5.2	1979-1981		44.1	1224	0.254		0.308

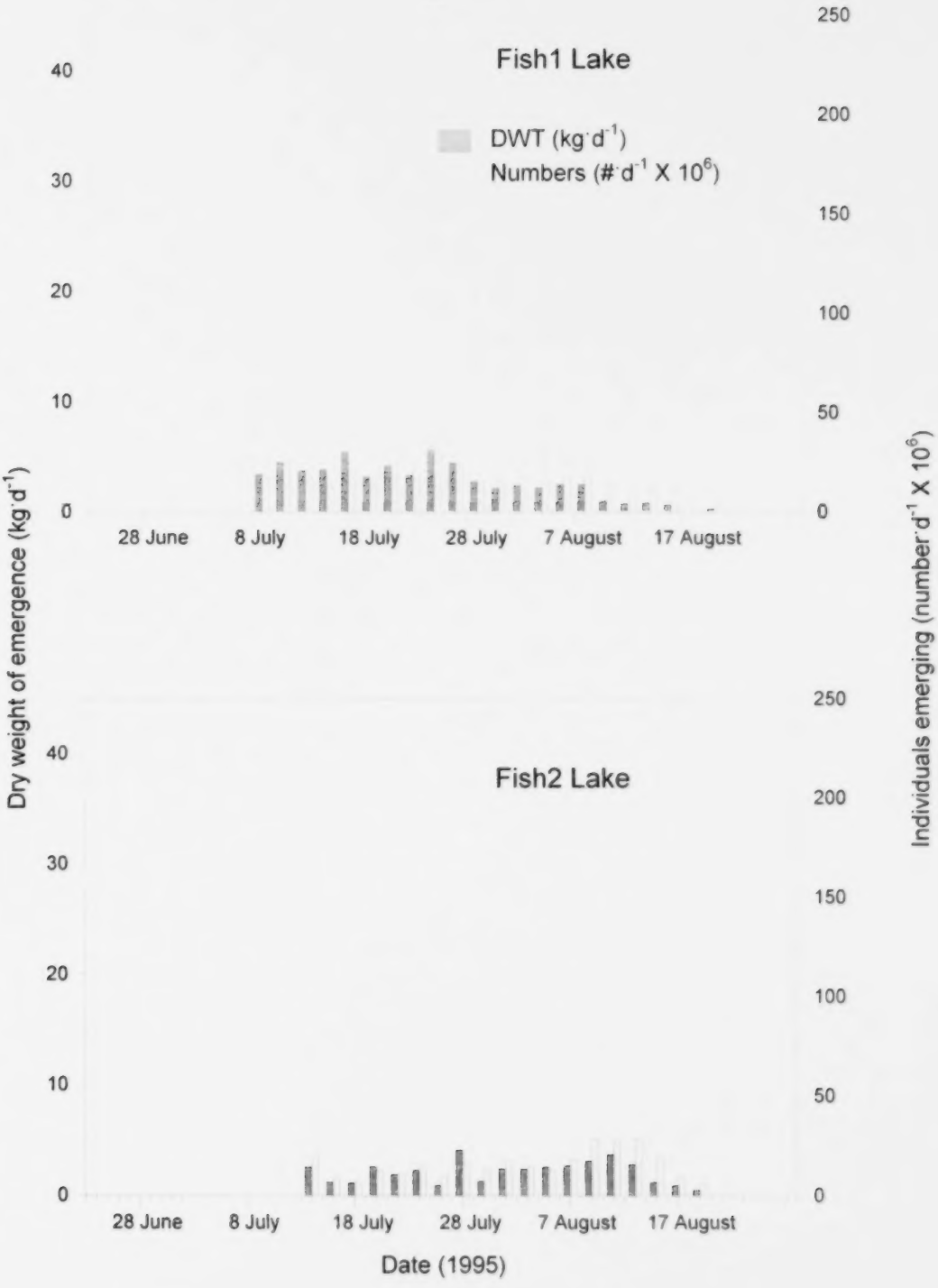

Fig. 11. Whole lake Diptera emergence for Fish1 and Fish2 lakes, 1995.

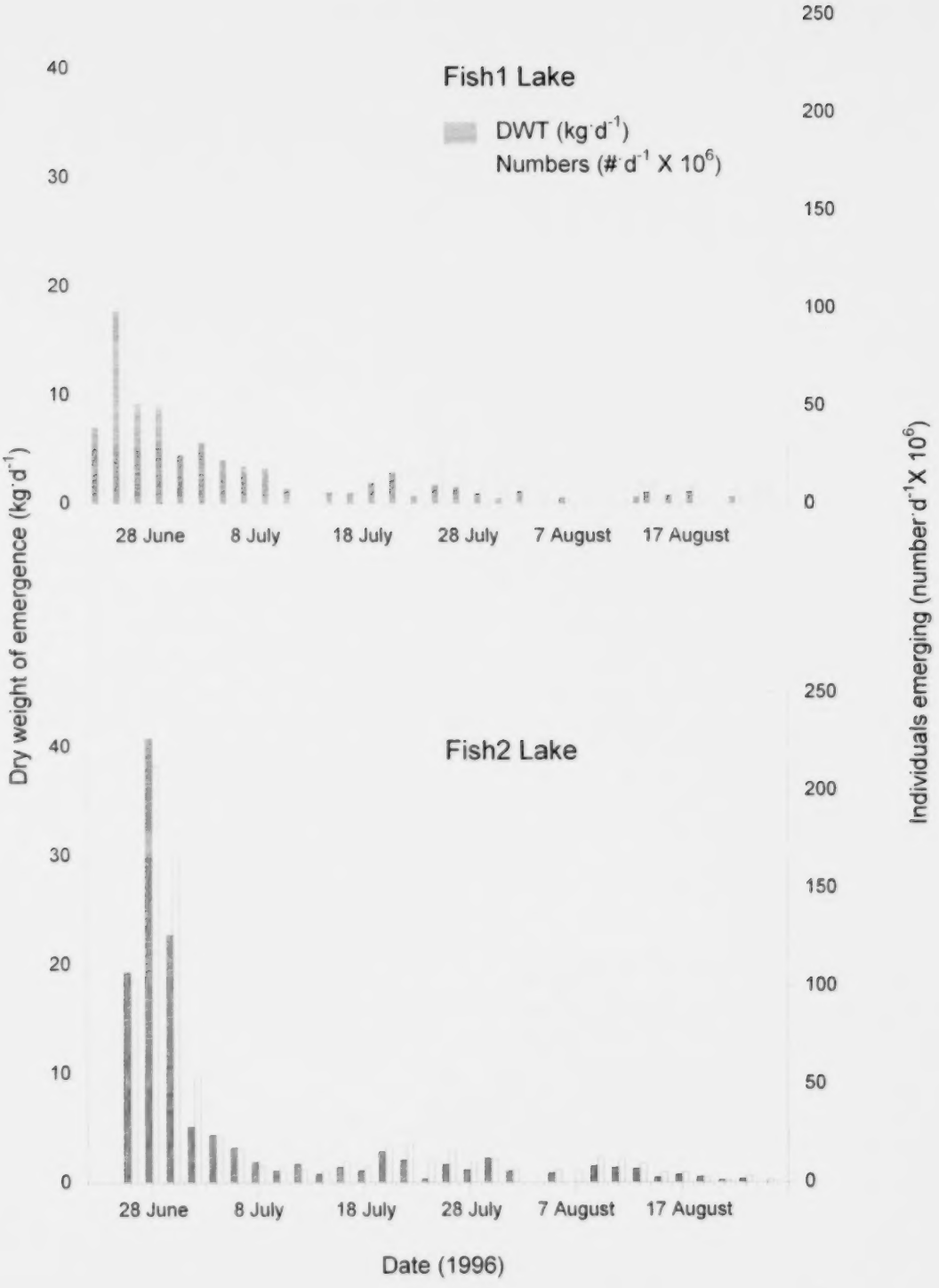

Fig. 12. Whole lake Diptera emergence for Fish1 and Fish2 lakes, 1996.

Capture success was related to gear type. Angling was used to successfully capture lake trout, and grayling were caught on flies. All species of fish were collected in trap nets, which had the best survival and was the only successful method found to capture burbot. All of the Salmonidae were caught by gillnets, but round whitefish in particular had difficulty recovering from this capture method. Summary capture statistics are presented in Table 9 for the fish from Fish1 and Fish2 lakes as part of a mark-recapture experiment. Small fish (< 15 cm) were not usually marked and were not included in this data unless the fish died. Mortalities in 1994 were largely a result of sampling fish for contaminant analysis. Burbot were not easily marked using the fin clip method employed for the other species, and as a result of this, population estimates based on mark-recapture were not made for this species, although the capture statistics were summarized (Table 9). Population estimates (Table 10) based on the mark-recapture experiment suggest that the adult fish community in Fish1 Lake is made up principally of round whitefish, followed by arctic grayling, and then lake trout. In Fish2 Lake the adult fish community is also dominated by round whitefish, then lake trout, followed by arctic grayling. The high abundance estimates and large standard errors for round whitefish result from the low number of recaptures of marked individuals. Recapture success for lake trout improved estimates of abundance.

Round whitefish and arctic grayling populations in both lakes were predominantly made up of smaller individuals, whereas lake trout populations tended to be dominated by larger individuals (Fig. 13 and 14). The importance of small fish was underestimated in this study. Small round whitefish, arctic grayling, and burbot were abundant in 1995 and 1996, and may have been in 1994, although no data were collected for fish < 15 cm in 1994. The large shallow areas with rocky substrate in both lakes and the vegetated areas particularly in Fish2 Lake may provide important habitat for rearing young fish (Appendix 1). In 1995, fish < 15 cm were counted and released with a general size range noted, however insufficient data were recorded to use in the length-frequency histograms. Fish1 Lake fish (< 15 cm) in 1995 included 311 arctic grayling, 619 burbot, 14 lake trout, and 154 round whitefish. Fish2 Lake fish (< 15 cm) in 1995 included 30 arctic grayling, 2 burbot, 65 lake trout, and 708 round whitefish. In 1996 many of the small fish were measured, and these data were included in the length-frequency histograms (Fig. 13 and 14). Fish1 Lake fish (< 15 cm), caught in 1996 included 349 arctic grayling, 7897 burbot, 6 lake trout, and 116 round whitefish. Fish2 Lake fish (< 15 cm) caught in 1996 included 80 arctic grayling, 47 burbot, 36 lake trout, and 1218 round whitefish. The small burbot collected were all young-of-the-year. The significant number of these small burbot suggest that burbot are a more important member of the fish community in these lakes than the capture of adults would indicate.

FISH DIET

Lake trout depended on zooplankton, Chironomidae and fish, which made up 86.6% of their diet (Fig. 15). Lake trout stomach contents (n=24, 2 were empty) were taken from fish ranging in size from 18.2 cm to 47.5 cm (average 38.6 ± 8.9 cm). The principal diet items included zooplankton (RI=66.1), Chironomidae (RI=11.8), and fish (RI=10.7). Lake trout were the only species of the three studied that had fish in their diet. Zooplankton were equally important in both July and August, Chironomidae were important in July and to a

Table 9. Fish mark-recapture summary information from Fish1 and Fish2 lakes, 1994 - 1996.

Lake	Species	Year	Total number captures	number Unmarked captures	number marked recaptures	new fish marked	number released no mark	1994 recaps	1995 recaps	1996 recaps	mortalities <10 cm	mortalities >10 cm
Fish1	burbot	1994	14	14	0	3	11	0			0	0
		1995	11	11	0	5	3	0	0		2	1
		1996	94	81	13	26	1	0	0	13	54	0
	whitefish	1994	125	123	2	97	0	2			0	26
		1995	91	85	6	71	4	1	5		2	9(1 recap)
		1996	280	269	11	220	0	0	2	9	8	41
	trout	1994	53	52	1	41	0	1			0	11
		1995	65	52	13	51	0	4	9		0	1
		1996	116	81	35	79	0	8	15	12	0	2
	grayling	1994	57	54	3	45	1	3				9
		1995	200	174	26	163	6	3	23		2	0
		1996	365	316	49	266	0	2	14	33	47	5
Fish2	burbot	1994	15	15	0	4	10	0			0	1
		1995	5	5	0	4	0	0	0		0	1
		1996	45	36	9	6	1	2	3	4	27	3(1 recap)
	whitefish	1994	54	54	0	20	1	0			0	33
		1995	77	75	2	48	7	0	2		7	14(1 recap)
		1996	503	494	9	238	82	0	1	8	119	57(2 recaps)
	trout	1994	174	173	1	146	4	1			0	22
		1995	251	203	48	180	16	15	33		1	7(2 recaps)
		1996	90	76	14	63	0	3	9	2		17(4 recaps)
	grayling	1994	30	30	0	17	1	0			0	12
		1995	55	41	14	35	0	1	13		1	8(3 recaps)
		1996	143	125	18	110	1	0	6	12	2	17(5 recaps)

Table 10. Population estimations for round whitefish, lake trout and arctic grayling in Fish1 and Fish2 lakes for 1994 and 1995 using the Jolly-Seber "death only" model. The 1996 estimate for Fish1 Lake lake trout is based on the Jolly-Dickson "death only" model with constant survival.

Lake	Species	Year	Abundance	Standard Error
Fish1	round whitefish	1994	15362.0	15208.4
		1995	4063.0	2269.8
		1996	-	-
		mean	9712.5	
	lake trout	1994	598.0	166.7
		1995	321.1	60.9
		1996	325.0	95.6
		mean	414.7	
	arctic grayling	1994	3706.4	1753.9
		1995	2212.3	486.6
		1996	-	-
		mean	2959.4	
Fish2	round whitefish	1994	6164.0	-
		1995	5353.5	5252.4
		1996	-	-
		mean	5758.8	
	lake trout	1994	2284.5	516.3
		1995	1089.5	274.2
		1996	-	-
		mean	1687.0	
	arctic grayling	1994	989.0	643.3
		1995	836.3	339.3
		1996	-	-
		mean	912.7	

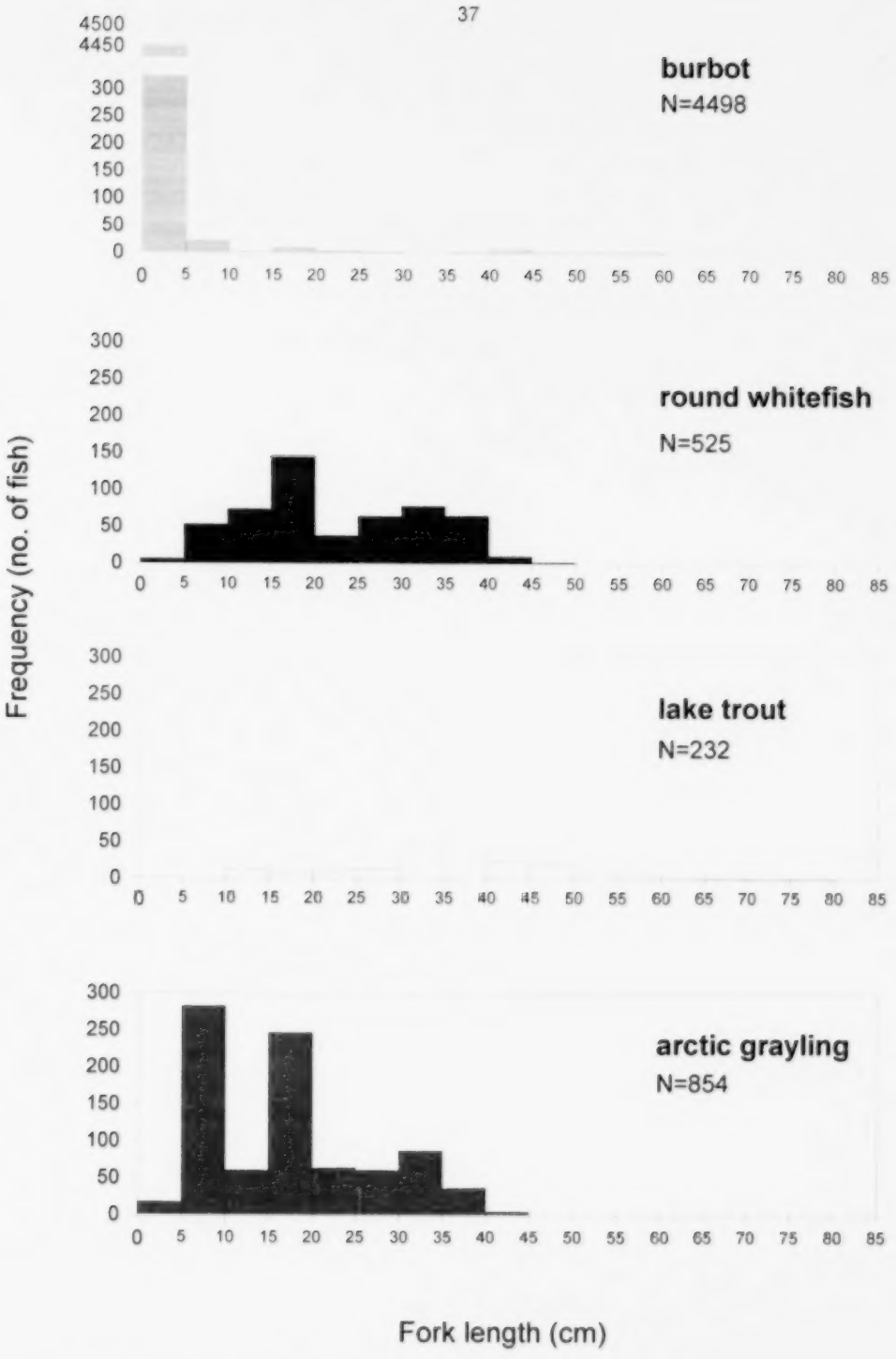

Fig. 13. Length-frequency distributions for Fish1 Lake fish, all years combined.

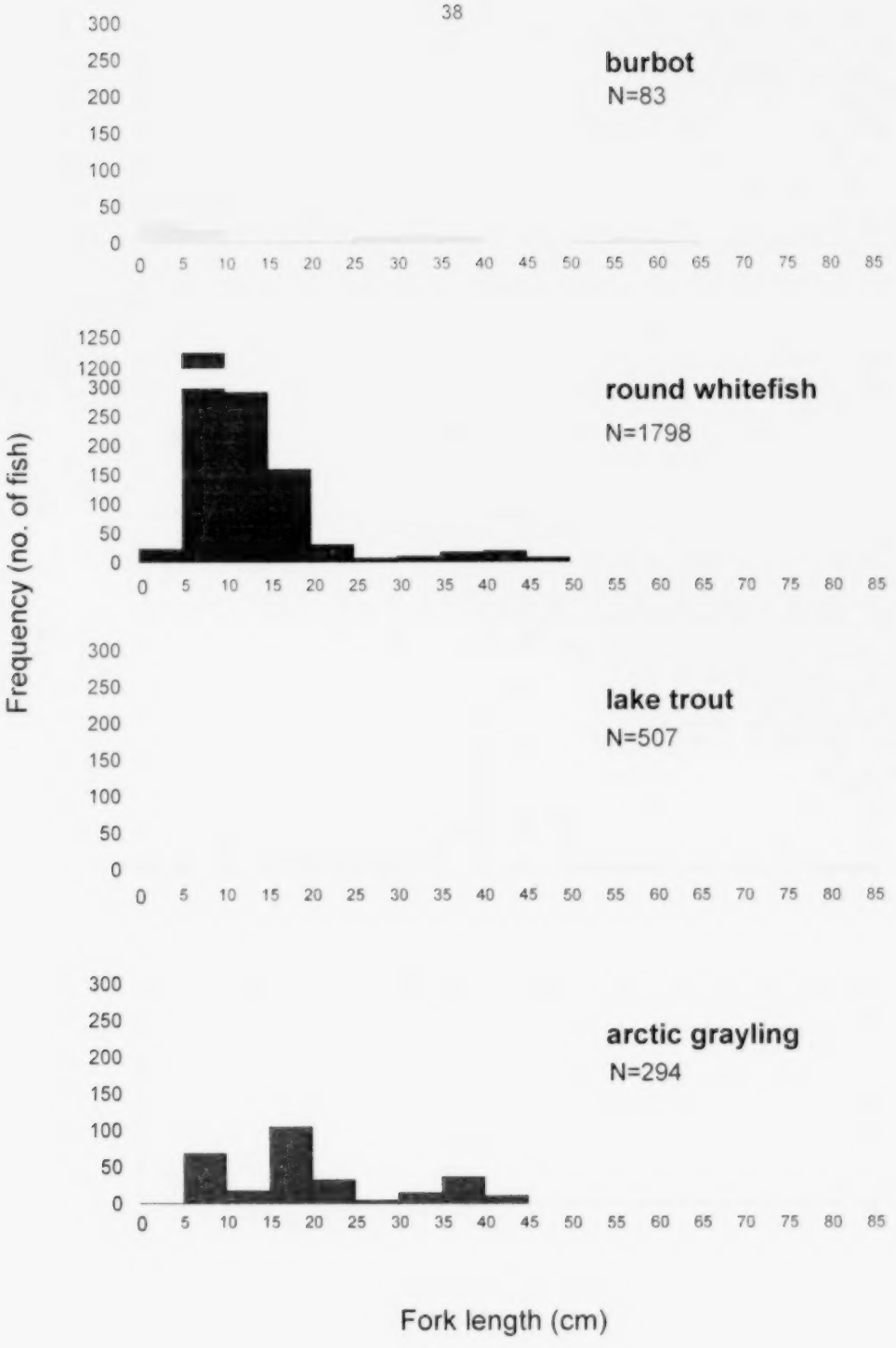

Fig. 14. Length-frequency distributions for Fish2 Lake fish, all years combined.

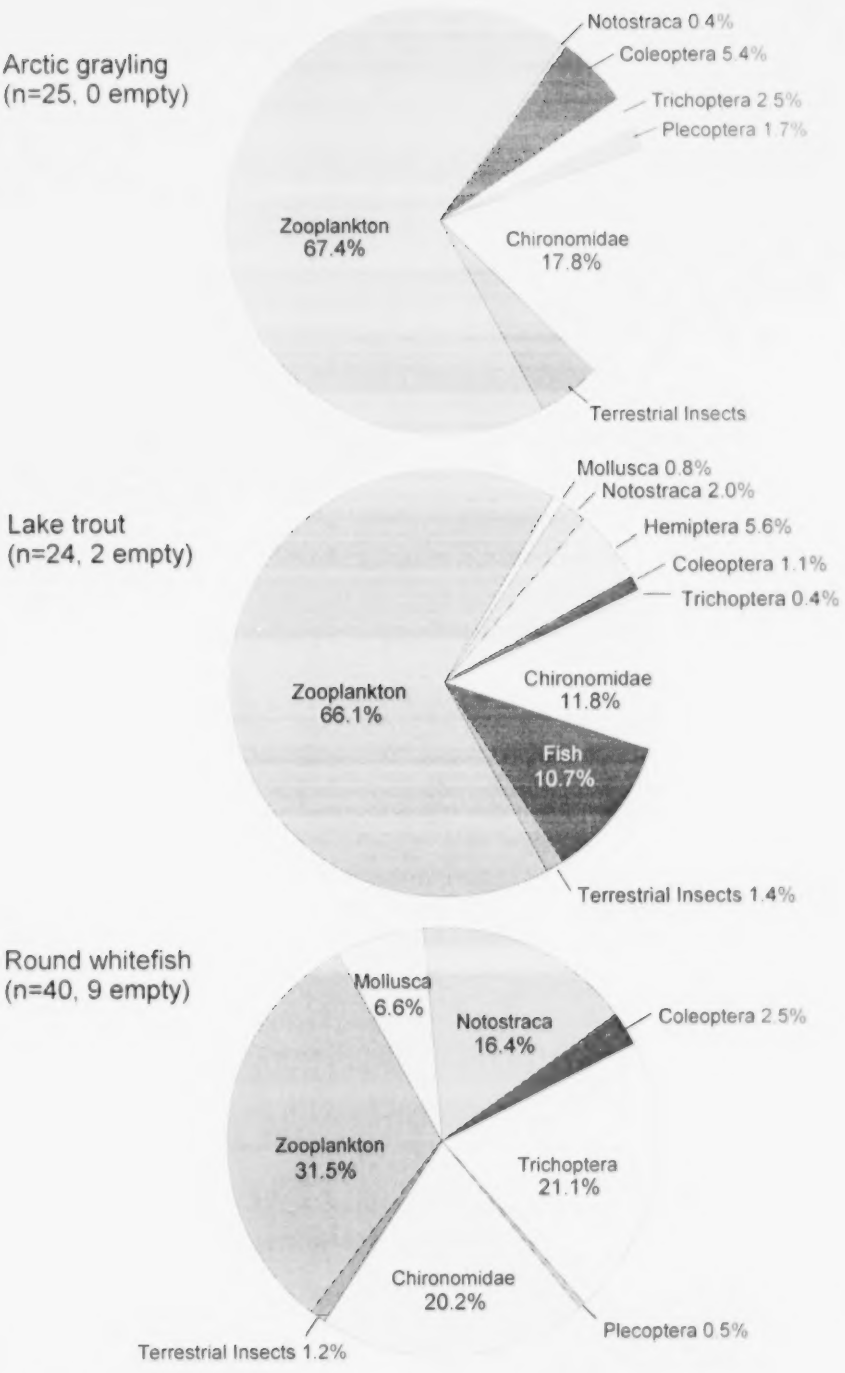

Fig. 15. Fish diet determined from stomach contents collected between June and August 1996 from Rea, Fish1 and Fish2 lakes combined. Results are summarized from Sotiropoulos (1997).

lesser extent in August, while the reverse held for fish remains (Sotiropoulos 1997). Lesser diet items included Hemiptera (RI=5.6), Notostraca (RI=2.0), terrestrial insects (RI=1.4), Coleoptera (RI=1.1), Mollusca (RI=0.8), and Trichoptera (RI=0.4). Hemiptera, Notostraca, Mollusca, and Trichoptera were only found in August (Sotiropoulos 1997). Coleoptera were only found in stomachs in July, while terrestrial insects, found in both months, were more important in July than August (Sotiropoulos 1997). Lake trout are generally considered to be predacious, feeding on a variety of invertebrates and small fish (McPhail and Lindsey 1970; Scott and Crossman 1973), and this was consistent with the lake trout sampled here. None of the lake trout sampled were exclusively piscivorous.

Arctic grayling diet was made up mainly of zooplankton and Chironomidae with these two items comprising 85.2% of their diet (Fig. 15). The arctic grayling (n=25, none were empty) sampled for stomach contents ranged in length from 11.6 to 42.5 cm (average 26.6 ± 9.6 cm). Zooplankton were the most important prey (RI=67.4), followed by Chironomidae (RI=17.8), Coleoptera (RI=5.4), terrestrial insects (RI=4.8), Trichoptera (RI=2.5), Plecoptera (RI=1.7), and Notostraca (RI=0.4). Zooplankton were less important in July than August, while the opposite held for Chironomidae (Sotiropoulos 1997). Chironomidae were principally pupae or emerging adults rather than larvae, and were likely taken while in the process of emerging. Plecoptera and Notostraca were only found in July (Sotiropoulos 1997). Trichoptera and terrestrial insects were found in stomachs collected in both months, but were more important in July than in August (Sotiropoulos 1997). Based on the literature, arctic grayling young feed mainly on zooplankton, gradually shifting to insects as they get larger (McPhail and Lindsey 1970). All sizes of grayling sampled as part of this study contained zooplankton. As adults, they consume a variety of aquatic and terrestrial insects, mollusks, occasionally fish, and even lemmings (McPhail and Lindsey 1970). Terrestrial insects were the most important summer food of grayling in several northern lakes, often comprising over half of the diet (McPhail and Lindsey 1970). This was not found in the arctic grayling sampled as part of this study (Fig. 15). However, grayling are considered opportunistic feeders, taking advantage of benthic taxa (e.g. amphipods in Great Slave Lake) (Rawson 1951) or caddis larvae and mayfly nymphs shortly after ice break-up, when little terrestrial food is available (Rawson 1950).

Round whitefish had a somewhat more varied diet with four major components: zooplankton, Trichoptera, Chironomidae, and Notostraca (*Lepidurus arcticus*), which comprised 89.2% of their diet (Fig.15). Round whitefish stomach samples (n=40, 9 were empty) were taken from fish ranging in length from 14.2 to 45.4 cm (average 28.8 ± 9.9 cm). As with the other species, zooplankton were the most important diet items (RI=31.5). Trichoptera (RI=21.1), Chironomidae (RI=20.2), Notostraca (RI=16.4), Mollusca (RI=6.6), Coleoptera (RI=2.5), terrestrial insects (RI=1.2), and Plecoptera (RI=0.5) were the remaining diet items in order of decreasing importance. Coleoptera, terrestrial insects, and Plecoptera were only found in July samples, while the remaining taxa were found in both months (Sotiropoulos 1997). Trichoptera and Chironomidae were equally important in both months, Notostraca were more important in August than in July, while the opposite held for zooplankton and Mollusca (Sotiropoulos 1997). Based on the literature, round whitefish are considered to be bottom-feeders, principally feeding in the shallow or inshore areas of lakes, eating a variety of benthic invertebrates. These include aquatic insects (caddis larvae and pupae, chironomid larvae), mollusks, and on occasion zooplankton, small fish, and fish eggs (McPhail and

Lindsey 1970; Scott and Crossman 1973). The diet of the round whitefish in this study is similar to this generalization however, with zooplankton being more important than an occasional diet item.

No stomach analyses were carried out on burbot in the study lakes. Based on the literature, they are considered to be voracious predators, consuming large quantities of aquatic invertebrates, insects, crustaceans, and mollusks when young, gradually adding fish to their diets until reaching a point when they become almost exclusively piscivorous (McPhail and Lindsey 1970; Scott and Crossman 1973). They were found to take advantage of the trap net to feed on young-of-year fish of all species.

TROPHIC STATUS

Some of the fish sampled in 1994 for contaminants were also analyzed for $\delta^{15}N$ and $\delta^{13}C$ (Fig. 16). Within a given lake, lake trout were positioned at the highest trophic level and arctic grayling at the lowest (Fig. 16). This is consistent with the stomach content data analyzed for this study and is a reflection of the importance of fish in the diet of lake trout. Smaller individuals may occasionally feed on small fish, but their importance in the diet usually increases as the lake trout grow. It is possible to find some lake trout, including some up to about 50 cm in length, feeding exclusively on invertebrates. However, many lake trout of this size have begun to rely increasing on fish as a major component of their diet. Beyond this size, lake trout usually switch to an entirely piscivorous diet feeding on larger and larger prey as they themselves grow. This would explain the high $\delta^{15}N$ levels found in the very large lake trout sampled (Fig. 17), particularly those from Lac de Gras (Fig. 16). The relationship between $\delta^{15}N$ and size for lake trout, (Fig. 17) ($\delta^{15}N = 7.808 + 0.00524 \cdot$ length, $R^2 = 0.473$, $P=<0.001$) may reflect the proportion of fish in the diet, and this in turn may be an indication of the availability of fish as a food source. Arctic grayling reliance on Chironomidae and zooplankton as the major components of their diet would result in their lower trophic position. The importance of both zooplankton and Chironomidae for these two species may explain the overlap in ^{13}C (Fig. 16). Round whitefish ^{15}N (Fig. 17) vary between the level found in arctic grayling and the lower levels found in some of the lake trout. Kidd et al. (1999) found that planktonic invertebrates were more depleted in ^{13}C and more enriched in ^{15}N when compared with most of the benthic invertebrates. Based on the results of Kidd et al. (1999), round whitefish with a broader range in diet items, including more benthic invertebrates, would be expected to result in a less depleted ^{13}C signal than either arctic grayling or lake trout, which was the case (Fig. 17). Although fish were not found in the round whitefish stomach samples analyzed in 1996, small fish and fish eggs (McPhail and Lindsey 1970; Scott and Crossman 1973) have been identified as part of their diets elsewhere. They may have been included in the diets of some of the larger round whitefish sampled for isotope analysis in 1994.

HABITAT

No attempt was made to describe habitats or their use by the fish within the lakes, however a brief survey using an electrofisher was made of two stream habitats connecting lakes in August 1994. The streams flowing into and out of the lakes provided habitat for the fish species along with access to different lakes (Appendix 1). The western inflow stream to Fish2 Lake was surveyed and ten arctic grayling, one lake trout, and ten sculpin were captured. The stream was made up largely of boulders with

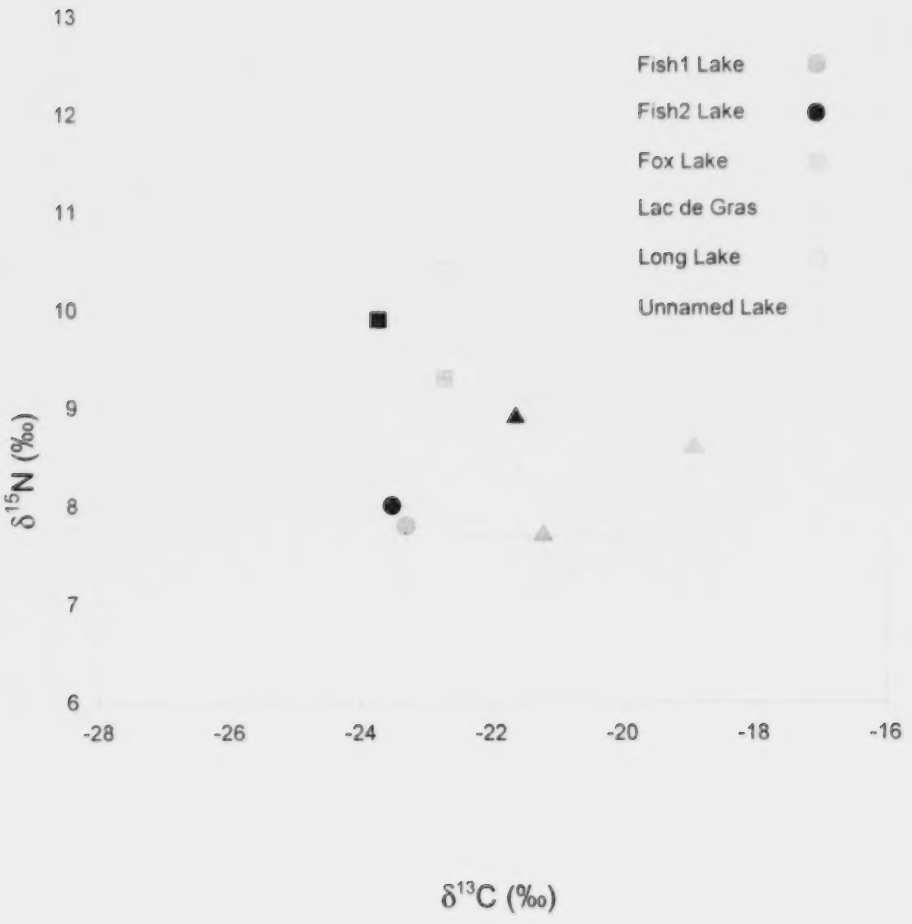

Fig. 16. Mean nitrogen ($\delta^{15}N$) and carbon ($\delta^{13}C$) stable isotopes (± SD) of arctic grayling (○), lake trout (□) and round whitefish (Δ) from each sampling lake.

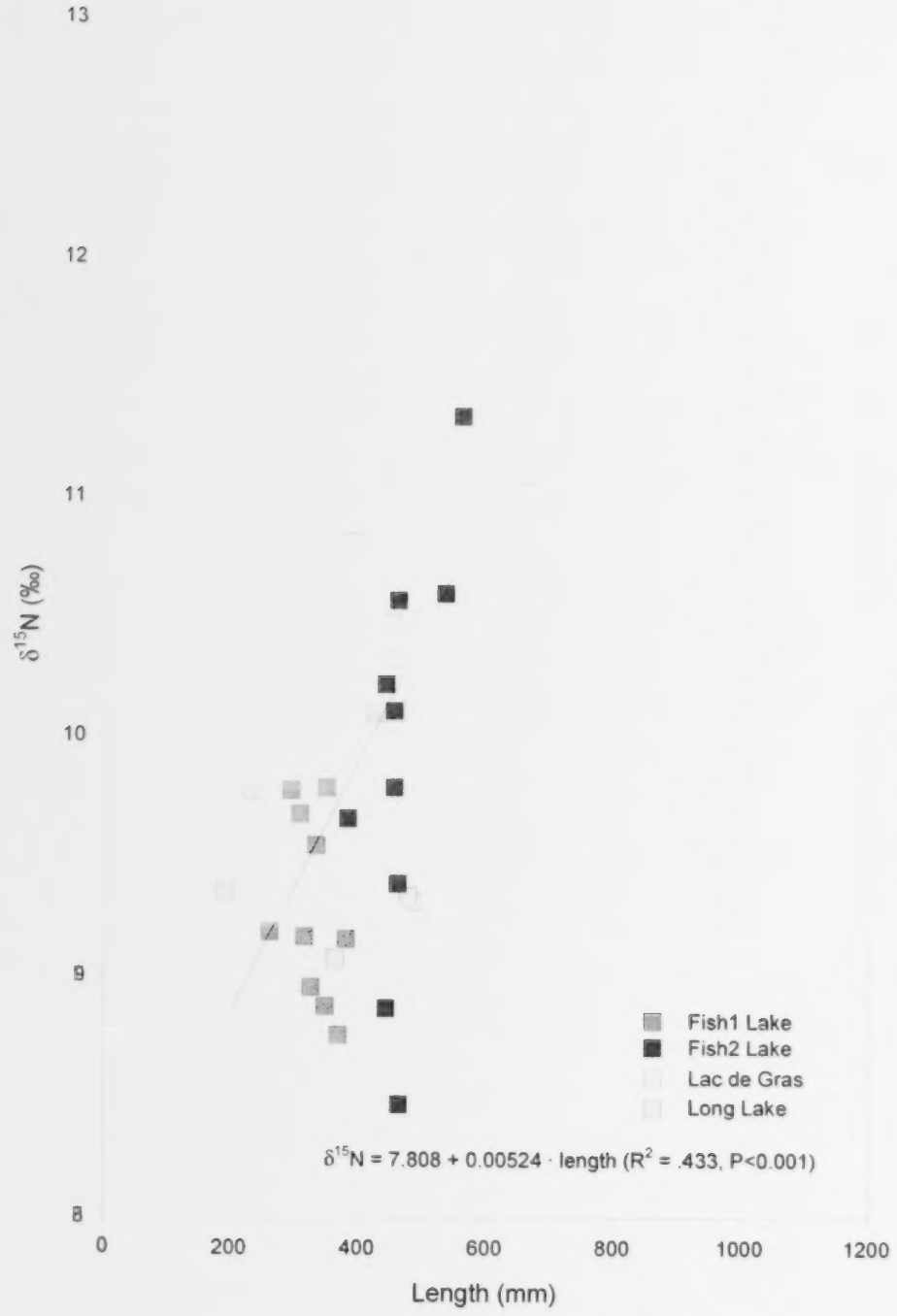

Fig. 17. Relationship between $\delta^{15}N$ and length for Lac de Gras area lake trout.

small areas of cobble, pebble, and gravel. The stream connecting Fish1 and Fish2 lakes was also surveyed. Eight arctic grayling, one lake trout, four burbot, and 26 sculpin were collected. The substrate in this stream was made up of a mixture of pebble, gravel, sand, and silt areas. The number of fish of each species was the only data recorded. The stream habitat may be used both by juveniles throughout the summer and by adults for spawning.

Although not found in the streams when surveyed in August, round whitefish are typically abundant in shallow areas of lakes and in clear streams (McPhail and Lindsey 1970). They spawn in late October in Northern Manitoba (Scott and Crossman 1973) along the shores of lakes or in streams (McPhail and Lindsey 1970). Harper (1948) reported upstream spawning migration in late October in the Northwest Territories. Round whitefish may access the stream habitat later in the fall, although there was no evidence that they moved between the lakes based on the mark-recapture data.

Lake trout, generally considered as inhabitants of deep cold lakes, are also abundant in shallow tundra lakes and large clear rivers (McPhail and Lindsey 1970). The trout captured during the stream survey indicate that they do make use of the stream habitats. Movement between the lakes was confirmed by one large (46.4 cm) lake trout which was first captured in Fish2 Lake on 27 June 1995. It was marked and released and was subsequently recaptured on 2 August 1995 in Fish1 Lake.

Arctic grayling spawn from April to June at about the time that lake ice-cover breaks up (McPhail and Lindsey 1970; Scott and Crossman 1973). Adults migrate from ice-covered lakes and larger rivers into small streams and spawn over gravel or rocky bottom (McPhail and Lindsey 1970; Scott and Crossman 1973). Young hatch within 16 -18 days at temperatures near 9ºC (McPhail and Lindsey 1970). Arctic grayling were found in one of the streams in August and may make use of this habitat over an extended period from the early spring spawning to fall. The streams do provide adult fish with access to the lakes in both directions. Seven adult grayling moved from one lake to the other during the course of the mark-recapture experiment. One moved from Fish2 Lake to Fish1 Lake, with the remainder moving in the opposite direction.

FISH CONTAMINANT RESULTS

Contaminants have been detected throughout the arctic at higher than expected levels (Jensen et al. 1997). Over the past decade, there has been a concentrated effort to increase knowledge of the levels of contaminants occurring in arctic fish (Muir et al. 1997). As part of this effort, information from the Lac de Gras area lakes was gathered in 1994 to add to the knowledge of levels and geographical variation of arctic fish contaminants, providing a baseline for future studies.

Metals Analysis

Trace metals were analyzed in fish tissue as part of this study. Ba, Be, Ca, Fe, Mg, Mn, PO_4, K, Na, Sr, and Sn were found above detection levels in the muscle of lake trout (Table 11). As, Cd, Cu, Pb, Hg, Se, and Zn were all found above detection limits by at least one of the laboratories (Table 12). This latter group of elements were analyzed in arctic grayling, lake trout, and round whitefish muscle tissue and all but Cd and Pb were also analyzed in liver tissue (Table 12). Sb, Mo, and Ag were below detection in this study, as well as in the Slave River Environmental Quality Monitoring Program (SREQMP) (Sanderson et al. 1997), which analyzed walleye, northern pike,

Table 11. Metals data from lake trout muscle samples from the Lac de Gras area. All measurements have been converted to $\mu g \cdot g^{-1}$ wet weight ± SD. Values below the detection limit (dl) are indicated by "<". The detection limit is used to calculate mean and standard deviations when some data were above and some below the detectable limit within a sample group.

Lake	% moisture	N	Al (dl=1.025)	Sb (dl=1.025)	Ba (dl=0.01)	Be (dl=0.002)	B (dl=0.103)	Ca (dl=0.103)	Cr (dl=0.308)	Co (dl=0.103)	Fe (dl=0.205)	Mg (dl=0.010)
Kodiak	83.3	1	<	<	0.017	<	0.484	128	<	<	5.711	197
Lac de Gras	76.7±2.6	4	<	<	0.016 ± 0.007	<	0.415 ± 0.038	137 ± 99	<	<	5.850 ± 2.055	229 ± 16
Long	81.9±1.6	5	<	<	0.013 ± 0.004	<	0.303 ± 0.064	101.1 ± 89.6	<	<	5.245 ± 1.476	206.1 ± 11.2
Nero	78.5±0.7	2	<	<	0.014 ± 0.005	<	0.226 ± 0.008	66.8 ± 8.9	<	<	3.610 ± 0.003	230.9 ± 6.1
Vulture	77.2	1	<	<	0.064	<	0.388	563	<	<	6.521	242

Lake	N	Mn (dl=0.021)	Mo (dl=0.205)	Ni (dl=0.051)	PO_4 (dl=2.050)	K (dl=0.103)	Ag (dl=0.205)	Na (dl=0.103)	Sr (dl=0.010)	Sn (dl=0.205)	Ti (dl=0.031)	V (dl=0.103)
Kodiak	1	0.075	<	<	6597	3969	<	423	0.29	0.484	<	<
Lac de Gras	4	0.093 ± 0.051	<	<	7410 ±395	4208 ± 361	<	271 ± 66	0.28 ± 0.24	<	<	<
Long	5	0.067 ± 0.010	<	<	6716 ± 477	3873 ±375	<	325 ± 335	0.23 ± 0.26	<	<	<
Nero	2	0.259 ± 0.270	<	<	7141 ± 236	4303 ± 237	<	146 ± 4	0.11 ± 0.02	<	<	<
Vulture	1	0.155	<	<	7843	3967	<	278	1.44	<	<	<

Table 12. Metals (sample means ± SD) analyzed by two different laboratories for fish muscle (M) and liver (L) sampled from the Lac de Gras area. All measurements have been converted to $\mu g \cdot g^{-1}$ wet weight. The detection limit (dl; Cantest, DFO when they differ) is used to calculate mean and standard deviations when some data were above and some below the detection limit within a sample group.

Species	Lake	Tissue	N	As (dl=0.01, 0.05)	Cd (dl=0.01, 0.0001)	Cu (dl=0.01)	Pb (dl=0.01, 0.03)	Hg (dl=0.001)	Se (dl=0.05)	Zn (dl=0.01)
lake trout	Fish1 (Cam)[1]	M	10	0.06 ± 0.03	0.0017 ± 0.0005	0.35 ± 0.16	0.06 ± 0.04	0.192 ± 0.186	0.44 ± 0.32	3.34 ± 0.39
	Fish2 (Ron)[1]	M	11	0.05 ± 0.05	0.0018 ± 0.0008	0.29 ± 0.07	0.06 ± 0.01	0.270 ± 0.121	0.36 ± 0.16	3.38 ± 1.01
		L	8	0.30 ± 0.11		22.42 ± 12.31		0.257 ± 0.142	1.18 ± 0.63	10.36 ± 4.19
	Kodiak[2]	M	1	<0.01	<0.01	0.46	<0.01	0.379		3.20
	Lac de Gras[2]	M	4	<0.01	<0.01	0.54 ± 0.20	<0.01	0.224 ± 0.037		3.76 ± 0.72
	Long[1]	M	19	0.05 ± 0.00	0.0010 ± 0.0001	0.25 ± 0.04	<0.03	0.346 ± 0.346	0.30 ± 0.07	3.65 ± 1.003
		L	5	0.12 ± 0.08		23.78 ± 13.49		0.417 ± 0.142	1.51 ±0.50	11.55 ± 4.51
	Long[2]	M	5	<0.01	<0.01	0.39 ± 0.14	<0.01	1.133 ± 1.072		3.60 ± 0.79
	Nero[2]	M	2	<0.01	<0.01	0.30 ± 0.09	<0.01	0.239 ± 0.114		4.00 ± 1.13
	Vulture[2]	M	1	<0.01	<0.01	0.42	<0.01	0.132		3.50
arctic grayling	Fish1 (Cam)[1]	M	5	<0.05	0.0016 ± 0.0007	0.34 ± 0.07	0.07 ± 0.08	0.076 ± 0.018	0.44 ± 0.36	4.68 ± 0.79
	Fish2 (Ron)[1]	M	9	<0.05	0.0007 ± 0.0006	0.27 ± 0.05	<0.03	0.093 ± 0.058		
		L	1	0.09		1.21		0.051	0.32	4.94
	Buster Lake[1]	M	1	<0.05	0.0010	0.27	<0.03	0.177	0.23	4.73
round whitefish	Fish1 (Cam)[1]	M	10	0.05 ± 0.01	0.0020 ± 0.0010	0.31 ± 0.07	<0.03	0.098 ± 0.034	0.67 ± 0.41	3.98 ± 0.99
	Fish2 (Ron)[1]	M	11	0.05 ± 0.01	0.0016 ± 0.0006	0.33 ± 0.06	0.04 ± 0.02	0.091 ± 0.043	0.70 ± 0.45	3.74 ± 1.38
		L	9	0.40 ± 0.23		9.93 ± 21.12		0.101 ± 0.098	0.88 ± 0.86	8.70 ± 2.38
	Fox[1]	M	1	<0.05	<0.0001	0.33	<0.03	0.066	0.58	4.40
	Long[1]	M	22	0.05 ± 0.01	0.0007 ± 0.0005	0.33 ± 0.12	<0.03	0.090 ± 0.070	0.45 ± 0.35	3.73 ± 1.05
		L	5	0.13 ± 0.09		1.88 ± 0.23		0.120 ± 0.142	0.78 ± 0.41	8.19 ± 2.12

[1] analyzed at DFO-FWI environmental chemistry laboratory, Winnipeg
[2] analyzed by Cantest Inc., Vancouver

lake whitefish, and burbot muscle, as well as burbot liver for 28 trace metals. Al, Be, Cr, Co, Ni, Ti, and Va were all below detection in this study, but were detected in fish analyzed as part of the Slave River Study (Sanderson et al. 1997). Ba, B, Ca, Fe, Mn, and Sr were all above detection levels in this study, but were at the bottom end of the Slave River Study range of values (Sanderson et al. 1997). Mg, PO_4, K, and Na were found within the range of values from the Slave River Study (Sanderson et al. 1997). Sn was found in levels above detection in a single lake trout from Kodiak Lake at nearly the same level as the single northern pike muscle sample from the Slave River Study above detection level (Sanderson et al. 1997). Sn was found at higher levels in burbot liver tissue from the SREQMP (Sanderson et al. 1997).

As is widespread in plant and animal tissues, but can be toxic in high concentrations (Förstner and Wittmann 1981). As was found above detection levels by the DFO laboratory, but not by Cantest (Table 12). The level found in muscle tissue is quite low. It was found in liver tissue at a higher level than in the muscle for all species (Table 12), indicative of accumulation in this organ. The concentrations are relatively low in comparison to levels found in muscle tissue of walleye, northern pike, lake whitefish, and burbot from the Slave River area (Sanderson et al. 1997). Five fish were analyzed in both labs. All five were below detection (dl=0.01 $\mu g \cdot g^{-1}$) in Cantest samples, four of five were below detection (dl=0.05 $\mu g \cdot g^{-1}$) DFO samples, and one of the five was 0.07 $\mu g \cdot g^{-1}$.

Cd can be toxic to freshwater aquatic life, and has properties intermediate with Zn and Hg, members of the same periodic sub-group (Förstner and Wittmann 1981). It was only measured in the muscle tissue of the three species here, although it is known to accumulate in the liver and kidneys (Köck et al. 1996). The levels found are similar to the levels reported in the Slave River study area for different species (Sanderson et al. 1997). The Slave study values were reported as being at the low end of values from other lakes in the Northwest Territories (Sanderson et al. 1997). Five fish were analyzed in both laboratories, and all were below detection (dl=0.01 $\mu g \cdot g^{-1}$ Cantest, dl=0.0001 $\mu g \cdot g^{-1}$ DFO).

Cu is an element essential to living organisms, but it can also be toxic at high levels, accumulating in the liver when excessive intake occurs (Förstner and Wittmann 1981). Muscle tissue levels found here in all three species were lower than the range of values reported from the Slave River study (Sanderson et al. 1997), and were similar to samples from other lakes in the Northwest Territories (Sanderson et al. 1997). The maximum value found in the liver in this study, however, was considerably higher than found in burbot liver (16.52 $\mu g \cdot g^{-1}$) from the Slave River study (Sanderson et al. 1997). The maximum value found in the liver came from a round whitefish in Fish2 Lake (61.98 $\mu g \cdot g^{-1}$), and several lake trout in Long and Fish2 lakes were found with levels as high as 37.68 $\mu g \cdot g^{-1}$ and 35.65 $\mu g \cdot g^{-1}$ respectively. Concentrations of Cu were comparable for all species (Table 12). A comparison of the five lake trout tissue samples analyzed by both laboratories was made. DFO lab results are on average 41.9% lower than the values determined by Cantest (range 19.4% - 57.6%). The high levels found in liver tissue suggest that the fish have been exposed to Cu. Fish2 Lake is in a drainage where no kimberlites have currently been discovered and has not undergone any lake-based drilling activities. Long Lake is in the Koala drainage, which does have kimberlite. Cu may be available through long-range transport, or may be available

through natural geological mineralization. Sediment core analysis would provide information useful in resolving sources of copper input.

Pb is a nonessential element that can be toxic when present in excessive amounts (Förstner and Wittmann 1981). Pb was below detection levels (dl=0.01 $\mu g \cdot g^{-1}$) in samples analyzed by Cantest, and as high as 0.32 $\mu g \cdot g^{-1}$ in a Fish1 Lake arctic grayling muscle tissue analyzed by DFO (dl=0.03 $\mu g \cdot g^{-1}$). This is slightly higher than the highest value from lake whitefish muscle in the Slave River study (Sanderson et al. 1997). No liver samples were analyzed, although Pb is known to accumulate in the liver and kidney (Köck et al. 1996). Concentrations for all three species were similar (Table 12). The five fish analyzed in both laboratories were all below detectable levels.

Se is essential for growth and fertility in animals, and for the prevention of diseases (Förstner and Wittmann 1981). It is also highly toxic in excessive amounts (Förstner and Wittmann 1981). Se was found to be between 0.04 $\mu g \cdot g^{-1}$ and 1.79 $\mu g \cdot g^{-1}$ in muscle tissue, and from 0.15 $\mu g \cdot g^{-1}$ to 3.07 $\mu g \cdot g^{-1}$ in liver tissue. DFO provided the only analysis for this metalloid, and it was not measured in the Slave River study. All three species sampled had similar concentrations (Table 12). Lemly (1996) recommends that tissue concentrations of Se in muscle, liver and ovary tissue of 8, 12, and 10 $\mu g \cdot g^{-1}$ dry weight respectively should be of concern (toxic effect thresholds for overall health and reproductive vigor of fish). This translates to 2 $\mu g \cdot g^{-1}$ wet weight in muscle tissue and 3 $\mu g \cdot g^{-1}$ wet weight in liver tissue. This would suggest that current levels are near or at levels of concern for some of the individual fish. Se may be available through long-range transport or may be available from the bedrock or soils in the area.

Zn is a trace metal of major importance in the metabolic functions of the cells that can be toxic when the nutritional supply becomes excessive (Förstner and Wittmann 1981). The range in values found here was similar to values found in the Slave River study. Liver values were higher, but similar to Slave River study fish. The highest value in liver tissue was 20.25 $\mu g \cdot g^{-1}$ from a lake trout in Fish2 Lake. Muscle values ranged from a minimum of 0.34 $\mu g \cdot g^{-1}$, to a maximum of 7.77 $\mu g \cdot g^{-1}$ from a Fish2 Lake round whitefish analyzed by the DFO laboratory. The three species sampled had similar concentrations (Table 12). Lake trout muscle tissue analyzed by Cantest ranged from 2.85 $\mu g \cdot g^{-1}$ to 4.93 $\mu g \cdot g^{-1}$ (Long Lake). Both labs analyzed five lake trout muscle samples, with the results from the DFO lab on average 11.3 % lower than the value determined by Cantest (range 2.4% - 24.3%).

Hg has been the predominant metal studied in arctic fish since early studies found Hg in arctic char (*Salvelinus alpinus*) (Smith and Armstrong 1975) and lake trout (Reinke et al. 1972). As fish are often important in the diet of local people and Hg is known to bioaccumulate in the aquatic food chain (Cabana et al. 1994), Hg continues to concern researchers. For most of the metals analyzed, concentrations in the liver were usually much higher than the levels found in muscle tissue, however this doesn't hold true in the case of Hg. The mean concentration of Hg in Long Lake's lake trout muscle tissue was 1.133 ± 1.072 $\mu g \cdot g^{-1}$ wet weight, which exceeds the guideline limit for commercial fish of 0.5 $\mu g \cdot g^{-1}$ and the recommended maximum for subsistence fish consumption of 0.2 $\mu g \cdot g^{-1}$ (Health and Welfare Canada 1984). Hg levels in arctic grayling and round whitefish were lower than the levels in lake trout, which may be explained by their lower trophic positioning (Fig. 15 and 16) (Kidd et al. 1998). The arctic grayling and round whitefish analyzed for metals were generally smaller and

younger on average than the lake trout. Five lake trout were measured by both laboratories. On average, DFO results were 19.8% lower than Cantest results for the same fish, however this ranged from 82.6% lower to 43.2 % higher.

Results for the metal concentrations from the same fish analyzed by two different laboratories differed for Cu, Zn, and Hg. This may have resulted from variability within the fish tissue itself, as metals are not necessarily accumulated homogeneously in the muscle tissue. White muscle differs from red muscle in metal concentrations. Analytical methodologies employed by both laboratories differed, however both are equally valid. As well, both laboratories followed QA/QC protocol and analyzed NRC reference materials along with the fish tissues. In the case of Cu and Zn, the readings reported by Cantest for the reference standards were above the 95% confidence interval in 3 out of 4 cases, which might explain the consistently higher values reported by Cantest versus DFO for these metals. Differences in the results of the Hg analyses were not consistent between labs, although both labs report the results of the reference standard analyses to be within the 95% confidence intervals for the certified values. This would suggest that the differences could be due to contamination or sample alteration/degradation during the field sampling, sub-sampling, handling, or storage of the tissue samples.

Metals are of concern when there is potential for excessive amounts entering the aquatic environment beyond the level that the system has become adapted to. Sources of metal pollution can originate from geological weathering, industrial processing of ores and metals, use of metals and metal components, leaching from garbage, and solid waste dumps, and input from animal and human excretions that contain heavy metals (Förstner and Wittmann 1981), and the long range atmospheric transportation of these pollutants. Mining operations in the Lac de Gras area could potentially redistribute elements from natural sources, thereby increasing their availability within the aquatic environment. Kimberlite is made up of a large list of elements, which have the potential to change the aquatic environment. Several, including Ba, Cerium (Ce), Cr, lanthanum (La), Mg, Niobium (Nb), and Ni, occur in higher concentration in kimberlite than in the bedrock of this region (Wilkinson et al. 2001). Harrison et al. (1995) were able to show effects of the mine effluents from the Lac de Gras diamond mines on fish under laboratory conditions. Kimberlite effluent composition was determined, but the source of the toxicity was unknown (Harrison et al. 1995). There is also the possibility of additional anthropogenic sources of trace metals as a result of the associated habitation and mining operations. Metals should be monitored in the area in the future. Organ tissue should be analyzed along with muscle tissue to better assess the biological availability of trace metals. Organ tissue generally has a greater affinity to heavy metals than does muscle tissue, making it more suitable for the evaluation of metal contamination and biological availability (Förstner and Wittmann 1981). Evaluation of gills for metals content and histological alteration might also provide useful information on exposure of fish to metals via uptake from the water. Burbot were not sampled for metals, although they occur in most of the lakes in the area. They should be sampled for contaminants including metals, as their trophic position would make them susceptible to metal bioaccumulation.

Liver Mixed Function Oxygenase (MFO) Enzyme Activity

Hepatic MFO (Mixed function oxygenase enzymes) analyses were carried out on lake trout, round whitefish, and arctic grayling collected from seven lakes in the area, including the two study lakes (Fish1 and Fish2). Results for EROD (ethoxyresorufin-*O*-deethylase) and AHH (aryl hydrocarbon hydroxylase) activity (reported in ηmol· mg protein^{-1} min^{-1}) and cytochrome P450 content (reported in ηm·mg·protein^{-1}) have been summarized in Table 13.

Within this study, CYP1A1 enzyme activity (as EROD) was found to be lowest in arctic grayling, higher in round whitefish, and highest in lake trout, when sexes were combined. Males did not show consistently higher EROD activity relative to females of the same species, as had been found by Williams et al. (1997). Arctic grayling were usually 0.001 or lower with a single value of 0.002. Round whitefish values ranged from a low in Fish1 Lake of 0.015 (± 0.012, N=10), to a high of 0.026 (± 0.019, N=11) in Fish2 Lake. Values in lake trout reached higher levels than those found in the other species. The lowest values for lake trout were in Fish1 Lake (0.012 ± 0.007, N=11); Fish2 Lake (0.020 ± 0.012, N=10); Nero Lake (0.040 ± 0.021, N=2); Long Lake (0.068 ± 0.050, N=14); Lac de Gras (0.084 ± 0.020, N=4); with the highest value found in Kodiak Lake (0.111, N=1). When compared to the activity levels in other studies, several of the lake trout in Lac de Gras area lakes had higher EROD activity levels than those found in Laberge and Kusawa lakes as reported by Muir et al. (1997).

Liver microsomal cytochrome P450 (CYP1A1) activity (as AHH) were similarly found to be lowest in arctic grayling, higher in round whitefish, and highest in lake trout from the lakes in this study when sexes were combined. Values from females were not generally lower than males (Table 13), in contrast to the findings of Williams et al. (1997). Arctic grayling values ranged from 0.020 (N=1) in Long Lake, to 0.031 (±0.011, N=8) in Fish1 Lake. For round whitefish, the lowest values were again from Long Lake 0.035 ± (0.017, N=8), and highest in Fish2 Lake at 0.057 (± 0.044,N=11). The values for whitefish AHH activity in this area are generally higher than those values reported for Laberge whitefish (Muir et al. 1997). Lake trout were the highest (Fish1 Lake 0.063 ± 0.019, N=11; Fish2 Lake 0.065 ± 0.023, N=10; Nero Lake 0.093, N=1; Lac de Gras 0.095 ± 0.008,N=4; Vulture Lake 0.124, N=1; Long Lake 0.138 ± 0.066, N=14; Kodiak Lake 0.198, N=1). The upper values from the lake trout are mid-way between the values from Lake Superior and Lake Ontario, as reported by Muir et al. (1997), and all values were higher than those found in Laberge and Kusawa Lake's lake trout (Muir et al. 1997). The lake trout AHH values were near or above those recorded in Saqvaqjuac area lakes (Lockhart 1995).

Cytochrome P450 content is also reported (Table 13). Unlike the relationship between species found with EROD and AHH activity, the relationship between species for cytochrome P450 content was not as clear. Values for P450 content for arctic grayling ranged from 0.228 (± 0.080, N=8) from Fish1 Lake, to 0.250 (± 0.051, N=6) for Fish2 Lake, higher than the values for round whitefish. Values from round whitefish ranged from 0.107 (± 0.041, N=8) from Long Lake, to 0.165 (± 0.058, N=11) from Fish2 Lake. Values for lake trout ranged from 0.183 (± 0.050, N=2) from Vulture Lake, to 0.272 (± 0.083, N=14) for Long Lake trout, which were similar to the values determined for arctic grayling. The content of P450 was also generally higher in males than in females.

Table 13. Mean (± S.D.) of mixed function oxygenase enzyme activity (ηmol· mg protein^{-1}·min^{-1} ± SD) as ethoxyresorufin-O-deethylase (EROD) and aryl hydrocarbon hydroxylase (AHH) and cytochrome P450 content (ηmol ·mg·protein^{-1}± SD) in liver from fish from the Lac de Gras area.

Lake	Species	Sex	N	EROD	AHH	P450
Fish1 (Cam)	lake trout	M	3	0.006 ± 0.008*	0.050 ± 0.017	0.250 ± 0.021[1]
		F	8	0.014 ± 0.006	0.067 ± 0.018	0.278 ± 0.068
	arctic grayling	M	1	0.001	0.028	0.370
		F	7	0.001 ± 0.000*	0.031 ± 0.012	0.208 ± 0.060
	round whitefish	M	6	0.020 ± 0.013	0.063 ± 0.050	0.134 ± 0.047[2]
		F	4	0.008 ± 0.004	0.025 ± 0.008	0.107 ± 0.078
Fish2 (Ron)	lake trout	M	7	0.019 ± 0.011	0.066 ± 0.025	0.253 ± 0.043
		F	3	0.022 ± 0.017	0.063 ± 0.023	0.195 ± 0.061
	arctic grayling	M	3	0.001 ± 0.000*	0.028 ± 0.003	0.272 ± 0.036
		F	3	0.001 ± 0.000*	0.025 ± 0.006	0.228 ± 0.061
	round whitefish	M	5	0.042 ± 0.015	0.099 ± 0.029	0.223 ± 0.023
		F	6	0.012 ± 0.004	0.023 ± 0.010	0.117 ± 0.012
Kodiak	lake trout	F	1	0.111	0.198	0.225
Lac de Gras	lake trout	M	1	0.055	0.083	0.244
		F	3	0.093 ± 0.009	0.099 ± 0.002	0.221 ± 0.038
	round whitefish	M	3	0.014 ± 0.003	0.043 ± 0.041	0.151 ± 0.014
		F	3	0.023 ± 0.011	0.034 ± 0.019	0.121 ± 0.014
Long	lake trout	M	8	0.056 ± 0.042	0.125 ± 0.042	0.304 ±0.046
		F	6	0.085 ± 0.058	0.157 ± 0.089	0.229 ± 0.106
	arctic grayling	M	1	0.002	0.020	0.254
	round whitefish	F	7	0.017 ± 0.008	0.036 ± 0.018	0.107 ± 0.041
Nero	lake trout	F	2	0.040 ± 0.021	0.093 ± 0.025	0.210 ± 0.033
Vulture	lake trout	F	2	0.073 ± 0.028	0.124 ± 0.027	0.183 ± 0.050

[1] no data for one sample
[2] no data for two samples
* The detection limit (0.001) is used to calculate mean and standard deviations when some data were above and some below the detection limits within a given sample.

Cytochrome P450 enzyme induction in liver has been the principal biomarker studied in arctic biota (Muir et al. 1997). Biomarkers are indicators of biological responses to contaminants, and are typically measures of normal processes that take on abnormal values as a result of exposure to chemicals of interest (Muir et al. 1997). The presence of functional cytochrome P450 enzymes (including EROD and AHH) means that OCs may be metabolized and eliminated, metabolized to lipophilic and toxic metabolites,and/or that OC exposure may lead to cytochrome P450 enzyme induction, increasing the amounts of metabolic enzymes present (de March et al. 1998). Cytochrome P450 mixed function oxygenase (MFO) enzymes are central to the metabolism of a wide array of xenobiotics (e.g. PAHs, and drugs) and endogenous compounds (e.g. hormones, steroids and fatty acids) (Williams et al. 1997). The enzymes act as detoxifiers by decreasing the lipid solubility of organic contaminants, thereby facilitating excretion (Williams et al. 1997), and as such are very sensitive indicators of exposure. MFOs have been shown to be indicators of oil pollution, PAH contamination, PCB incineration, pulp mill effluent discharges, and polluted river systems (Williams et al. 1997). The limited amount of work on biomarkers in arctic animals, however, means that the range of normal responses for these indicators have not been well defined (Muir et al. 1997). The relatively high levels of AHH found in some of the fish in the Lac de Gras area (particularly the lake trout from Long and Vulture lakes) are higher than levels found in any of the arctic samples reported in Muir et al. (1997). They are in between the levels found in Lake Superior lake trout and Lake Ontario lake trout, which are exposed to higher amounts of a variety of chemical contaminants (Muir et al. 1997). The levels may be within the range of normal responses for arctic fish, or may indicate exposure to some form of contaminant. Muscle tissue PAHs were measured on some of the fish analyzed for MFOs, however there was no correlation between the two. If samples are analyzed for MFOs in the future, tissue from the same fish should also be analyzed for other persistent organic pollutants (POPs) to determine if correlations exist. It is also interesting to note that the level of EROD and AHH activity for the three species sampled mimics the trophic relationship described by stable isotope analysis. Arctic grayling generally being lowest, lake trout highest and round whitefish intermediate between the two (Fig. 18).

Organochlorine (OC) Contaminants: Toxaphene, Polychlorinated Biphenyls (PCBs), Dioxins and Furans

In the past three decades, it has been recognized that organochlorine compounds have been reaching the arctic and accumulating in ecosystems (Barrie et al. 1997). Elevated concentrations of these compounds in the tissues of marine mammals and fish, especially OC pesticides, can only be explained by sources outside of the Arctic (Barrie et al. 1997). Most OCs are human-made, rather than being derived from natural sources (Barrie et al. 1997), and they include organic pesticides and chlorinated industrial organic compounds. Organochlorine pesticides have been used on a global scale since the 1950s (Barrie et al. 1997). Dioxins and furans are formed as by-products of other processes, such as incineration, and have toxicological effects in the environment at very low concentrations (Barrie et al. 1997). Sixteen lake trout from four lakes in the Lac de Gras area were analyzed for a suite of organochlorine compounds. Biological data for individual fish are summarized in Table 14. Organochlorine pesticide

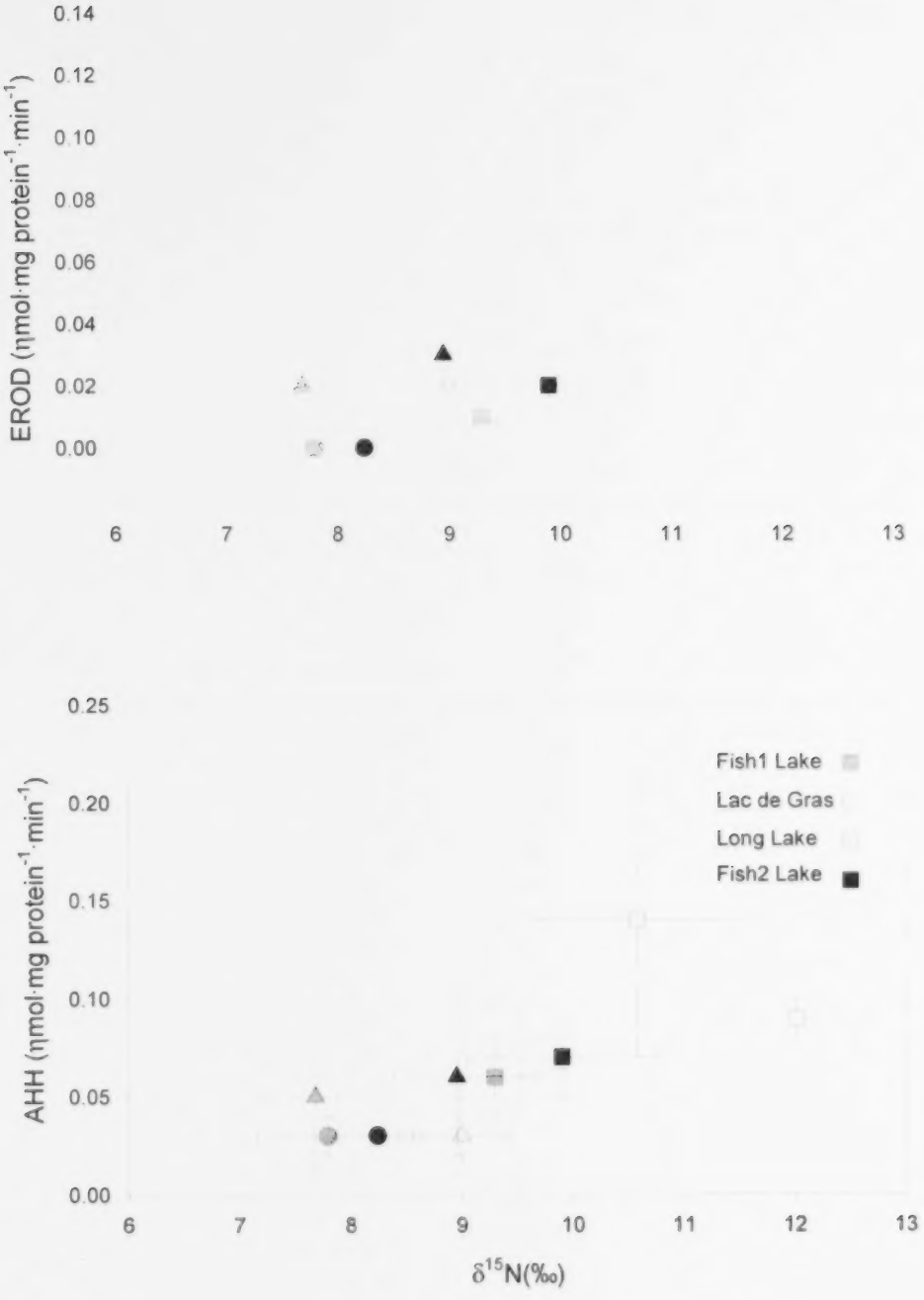

Fig. 18. Relationship between mean EROD and AHH activity (±SD) and nitrogen (δ^{15}N) stable isotope (±SD) for arctic grayling (○), lake trout (□), and round whitefish (Δ) from several Lac de Gras area lakes.

Table 14. Fish sample data corresponding to contaminant samples analyzed for organochlorine compounds (PCB, DDT, toxaphene, etc.) and dioxin/furan isomers. All are lake trout from the Lac de Gras area. The analysis is based on whole fish portions.

Lake	Date	Sample	Length (mm)	Weight (g)	%lipid (whole)	%lipid (fillet)	Sex
Grizzly	25-Feb-94	1	490	1099.5	3.27	1.09	F
		2*	450	848.1	4.01	1.11	M
		3*	500	1621.1	5.92	1.72	F
		4	550	1308.9	4.57	1.02	F
Long	20-Jan-94	5	530	1095.1	2.01	0.88	F
		6*	180	68	-	-	Imm
		7	300	113.5	-	-	Imm
		8*	350	350.6	1.17	0.92	M
		9	570	1543.2	1.88	0.99	M
		10	550	1701.8	2.32	1.2	M
		11*	350	351.8	1.51	0.61	M
Nero	05-Feb-94	12	630	2209.2	2.9	0.93	M
		13	610	1780.2	3.67	1.21	M
		14	500	1004.2	4.48	1.3	M
		15*	370	520.5	4.16	1.39	M
Vulture	10-Mar-94	16*	550	1476.6	13	6.93	M

* indicates duplicate samples analyzed

results (expressed in $\mu g \cdot g^{-1}$ wet weight) are presented in Table 15 for each fish individually. Many of the compounds were at or below detection levels.

Hexachlorocyclohexane (HCH) is a pesticide made up of a mixture of isomers, most of which have been banned from use in the U.S. and most other circumpolar countries since the late 1970's, although they are still used in China. The γ-HCH isomer (Lindane) is the only form of HCH currently used in its pure form in North America, Japan, and Europe, where it is used in seed treatments. All fish sampled for β-, γ-, and δ-HCH were below detection (Table 15). Three of the 16 fish sampled for α-HCH had levels higher than the detection limit (Table 15), and the values ($<0.002 - 0.005 \mu g \cdot g^{-1}$) were within the range of those found in the Slave River study (Sanderson et al. 1997). They were at the high end of the range of values reported for *Salvelinus namaycush* in lakes across northern Canada (de March et al. 1998).

Octachlorostyrene is a persistent, bioaccumulative, and toxic substance, which is formed as a by-product of high temperature industrial processes involving chlorine. It has no natural sources and is not produced commercially. Release to the environment occurs in effluent from chlorine and gas production, aluminum smelting, and other metal production. It has also been found in leachate from industrial landfills and fly ash from waste incinerators. Five of the 16 fish analyzed for octachlorostyrene were above detection, with the highest concentration 0.006 $\mu g \cdot g^{-1}$ wet weight. This is higher than the 0.002 $\mu g \cdot g^{-1}$ maximum level found in the Slave River study (Sanderson et al. 1997).

Chlordane (CHL) is a manufactured chemical that was used as a pesticide in the United States from 1948 to 1988, when the U.S. Environmental Protection Agency (EPA) banned all uses. Technical grade chlordane is a mixture of at least 120 compounds. Three of the sixteen fish tested had detectable levels of α-chlordane (up to 0.007 $\mu g \cdot g^{-1}$), while γ-chlordane was below detection for all fish sampled. The range for α-chlordane fell within the range of concentrations from fish sampled as part of the SREQMS (Sanderson et al. 1997) and from elsewhere across northern Canada (de March et al. 1998).

Aldrin and dieldrin are two closely related insecticides. They were widely used as agricultural insecticides, veterinary agents, termiticides, and vector control agents from the 1950s to the early 1970s. Aldrin is readily converted to dieldrin in the environment. Although the use of aldrin and dieldrin has been severely restricted or banned in manyparts of the world since the mid-1970s, they are still used in termite control in some countries. Aldrin was below detection in all fish sampled (Table 15). Dieldrin was above detection in one fish (0.006 $\mu g \cdot g^{-1}$), at detection levels in three fish, and below detection for the remainder (Table 15). This was within the range of concentrations found in the Slave River study (Sanderson et al. 1997).

DDT (dichlorodiphenyltrichloroethane) is an insecticide used mainly to control mosquito-borne malaria and as an agricultural insecticide. DDT was banned for use in the U.S. in 1972, although it is still used in other (primarily tropical) countries. Technical grade DDT is actually a mixture of three isomers of DDT, principally the *p,p'*-DDT isomer, with the *o,p'*-DDT and *o,o'*-DDT isomers typically present in much lesser amounts. DDT and related pesticides were found in low concentrations in a number of fish. The degradation product *p,p'* DDE was detected in three fish at 0.002 $\mu g \cdot g^{-1}$ and one fish at 0.006 $\mu g \cdot g^{-1}$, and *p,p'* DDD was found at the detection limit in a single fish (Table 15). The parent compounds were detected in five fish at or above detection

Table 15. Organochlorine Residues ($\mu g \cdot g^{-1}$ wet weight) data from lake trout samples (whole fish portions) collected in the Lac de Gras area early in 1994. Sample numbers correspond to individual fish data summarized in Table 14.

Sample Number	α-HCH	β-HCH	γ-HCH	δ-HCH	Octachloro-styrene	α-Chlordane	γ-Chlordane	Dieldrin	Aldrin	*p,p'* DDE	*p,p'* DDD	*o,p'* DDT	*p,p'* DDT	ΣDDT[1]
1	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	0.002	0.002
2	0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	0.002	<0.002	0.002	<0.002	<0.002	<0.002	0
2*	-	-	-	-	-	-	-	-	-	-	-	-	-	-
3	0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	0.002	<0.002	0.002	<0.002	<0.002	<0.002	0
3*	0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	0.002	<0.002	0.002	<0.002	<0.002	<0.002	0
4	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	0.002	<0.002	0.002	<0.002	<0.002	<0.002	0
5	<0.002	<0.002	<0.002	<0.002	0.002	0.002	<0.002	<0.002	<0.002	<0.002	<0.002	0.003	0.003	0.003
6	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002
6*	-	-	-	-	-	-	-	-	-	-	-	-	-	-
7	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	0
8	<0.002	<0.002	<0.002	<0.002	0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	0
8*	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	0
9	<0.002	<0.002	<0.002	<0.002	0.003	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	0.002	0.002	0.002
10	<0.002	<0.002	<0.002	<0.002	0.006	0.003	<0.002	<0.002	0.002	<0.002	0.002	0.005	0.005	0.005
11	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	0
11*	-	-	-	-	-	-	-	-	-	-	-	-	-	-
12	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	0
13	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	0
14	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	0
15	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	0
15*	-	-	-	-	-	-	-	-	-	-	-	-	-	-
16	0.005	<0.002	<0.002	<0.002	0.003	0.007	<0.002	0.006	<0.002	0.006	<0.002	0.006	0.007	0.007
16*	0.005	<0.002	<0.002	<0.002	0.002	0.006	<0.002	0.005	<0.002	0.005	<0.002	0.005	0.005	0.005

*indicates analysis of duplicate samples

[1]ΣDDT includes the total of *p,p'* DDE, *p,p'* DDD, *o,p'* DDT, *p,p'* DDT and *o,p'* DDE.

Table 15 cont'd. Organochlorine Residues ($\mu g \cdot g^{-1}$ wet weight) data from lake trout samples (whole fish portions) collected in the Lac de Gras area early in 1994. Sample numbers correspond to individual fish data summarized in Table 14.

Sample Number	HCB	Heptachlor	Heptachlor Epoxide	Endrin	beta Endosulfan	Methoxychlor	Mirex	Photomirex	Total PCB	Toxaphene
1	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	0.090	<0.03
2	0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	0.056	<0.03
2*	-	-	-	-	-	-	-	-	-	<0.03
3	0.003	<0.002	<0.002	<0.002	<0.002	<0.002	0.002	0.002	0.045	<0.03
3*	0.004	<0.002	<0.002	<0.002	<0.002	<0.002	0.002	<0.002	0.051	-
4	0.003	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	0.002	0.075	<0.03
5	0.003	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	0.110	0.051
6	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	0.039	<0.03
6*	-	-	-	-	-	-	-	-	-	<0.03
7	0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	0.048	<0.03
8	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	0.076	<0.03
8*	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	0.069	-
9	0.004	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	0.072	0.063
10	0.005	0.002	<0.002	<0.002	<0.002	<0.002	0.004	0.003	0.110	0.114
11	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	0.056	<0.03
11*	-	-	-	-	-	-	-	-	-	-
12	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	0.035	<0.03
13	0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	0.047	<0.03
14	0.003	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	0.062	<0.03
15	0.003	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	0.058	<0.03
15*	-	-	-	-	-	-	-	-	-	0.286
16	0.015	<0.002	0.003	<0.002	<0.002	<0.002	0.003	0.003	0.110	0.192
16*	0.014	<0.002	0.003	0.002	0.002	<0.002	0.004	<0.002	0.160	-

*indicates analysis of duplicate samples

levels (0.002-0.006 $\mu g \cdot g^{-1}$), and Sum-DDT ranged from 0.002 – 0.006 $\mu g \cdot g^{-1}$ in five fish (Table 15). The samples from the SREQMS had a higher range of values for *p,p'* DDE, the same range for *p,p'* DDD, and slightly lower ranges for the parent compounds (Sandersen et al. 1997). The total DDT values found here were within and near the bottom of the range of values from lakes across northern Canada (de March et al. 1998).

Hexachlorobenzene (HCB) is a persistent, bioaccumulative, and toxic compound widely used as a fungicide until 1965. It was also used to make fireworks, ammunition, and synthetic rubber. Currently, there are no commercial uses of HCB, but it is still found as by-products, in the manufacture of other chemicals, chemical solvents, incineration of waste products, and contamination in other pesticides. Levels found in this study ranged from below detection to 0.015 $\mu g \cdot g^{-1}$ wet weight. Most fell within the range of concentrations found in the SREQMS, although the highest concentration from the Vulture Lake lake trout was near the top end of the concentrations found in liver tissue in the SREQMS (Sanderson et al. 1997).

Heptachlor was used extensively in the past for killing insects in homes, buildings, and on food crops. Its use slowed in the 1970s and was severely restricted in 1988. Heptachlor epoxide is a breakdown product of heptachlor and is more likely to be found in the environment than Heptachlor. Heptachlor is present as an impurity in the pesticide chlordane. Heptachlor was at detection levels in one fish and below for the remainder in this study, whereas it was not detected in any samples in the SREQMS (Sanderson et al. 1997). Heptachlor expoxide was found above detection levels in the single lake trout from Vulture Lake (0.003 $\mu g \cdot g^{-1}$ wet weight), which is within the range found in the SREQMS (Sanderson et al. 1997).

Endrin has been used as a pesticide to control birds on buildings and insects and rodents in fields and orchards. First introduced in 1951, many countries, including the U.S., banned the use of endrin in the 1980s. However, many other countries continue to permit the import and use of endrin. All fish samples were below detection except the Vulture Lake lake trout, which was at the detection level (Table 15). The values here were within the range found in the SREQMS (Sanderson et al. 1997).

Beta endosulfan is an insecticide used to control insects on grains, tea, fruits, vegetables, tobacco, and cotton. In the United States, endosulfan is mainly applied to tobacco and fruit crops. It is also used as a wood preservative. Endosulfan is sold as a mixture of two different forms of the same chemical (alpha- and beta-endosulfan). It has not been produced in the United States since 1982; however, it is still used in the U.S. to produce other chemicals. All fish samples were below detection, except the Vulture Lake lake trout, which was at the detection level (Table 15). The values here were within the range found in the SREQMS (Sanderson et al. 1997).

Methoxychlor is used against a wide variety of insects, including flies, mosquitoes, and cockroaches. It is used on agricultural crops, livestock, animal feed, grain storage, home gardens, and on pets. Methoxychlor is one of a few organochlorine pesticides that have seen an increase in use since the ban on DDT in 1972. However, early in 2000, the U.S. EPA issued an order preventing further manufacture and sale of the product by the manufacturer, although existing stocks can still be used. All fish samples were below detection (Table 15). Methoxychor was only detected in one burbot liver at 0.002 $\mu g \cdot g^{-1}$ in the SREQMS (Sanderson et al. 1997).

Mirex was used in insect bait formulations and was used extensively in the southeastern U.S. to control fire ants beginning in 1958. In 1977, the U.S. EPA cancelled the registrations of pesticides containing this chemical, and it was banned in 1979. Mirex was also used as a flame retardant in plastics, rubber, paint, paper, and electrical goods. Mirex is considered a stable, bioaccumulative, and persistent pesticide, which is an endocrine disruptor. In the presence of light, mirex undergoes photochemical conversion to photomirex. Whole fish samples were analyzed for both mirex and photomirex. Two fish were at detection levels, two were at 0.004 $\mu g \cdot g^{-1}$ wet weight, and the remainder were below detection (Table 15). This is slightly above the range detected in any of the whole fish sampled as part of the SREQMS (Sanderson et al. 1997). Photomirex was found at 0.003 $\mu g \cdot g^{-1}$ wet weight in two fish, at detection in two others, and below detection for the remainder (Table 15), which is within the range found as part of the SREQMS (Sanderson et al. 1997).

Polychlorinated biphenyls (PCBs) are industrial OCs, which were first produced in 1929 and were used as hydraulic and transformer fluids. Their production and use has been largely discontinued in western countries (Barrie et al. 1997). All values for ortho-substituted congeners (28, 52, 89/101, 137, 138, 153, 170 and 180) were below detection limits of 0.01 $\mu g \cdot g^{-1}$. As part of the SREQMS, the same congeners were found above detection limits in walleye samples (except PCB 118, which was not detected), and in northern pike and lake whitefish only PCB 105 was detected (Sanderson et al. 1997). Liver tissue was analyzed from SREQMS, and all congeners were detectable (Sanderson et al. 1997). Results for coplanars, considered to be the most toxic of the PCBs ($pg \cdot g^{-1}$ wet weight), are summarized in Table 16. The concentrations found here were higher than the range from SREQMS for PCB 77, PCB 105, PCB 126, and PCB 169 (Sanderson et al. 1997). Coplanar PCB 81 was higher in the Slave River lake whitefish, although the remaining samples were all lower than found here (Table 16) (Sanderson et al. 1997). The mean concentrations expressed as $ng \cdot g^{-1}$ wet weight for total PCBs are: Grizzly Lake 67.3 ± 18.9, Long Lake 72.5 ± 28.3, Nero Lake 50.5 ± 12.1, and Vulture Lake 135. The values were comparable to the range found in walleye from SREQMS, Slave River northern pike, and lake whitefish (Sanderson et al. 1997). The Vulture Lake concentration was outside of the range found in SREQMS for any fish, although lower than the liver samples analyzed (Sanderson et al. 1997). These values are within the range of values reported in Muir et al. (1997) and de March et al. (1998) for lake trout from lakes in Yukon Territory, Northwest Territories, and northern Quebec. They were considerably lower than values for Lake Laberge lake trout and two other Yukon lakes, but higher than most of the other lakes sampled (Muir et al. 1997, de March et al. 1998). Unlike the other reported studies (Muir et al. 1997), total PCB levels were higher than toxaphene in the four lakes sampled here (Table 15).

Toxaphene is a complex mixture of polychlorobornanes and camphenes, widely used in the U.S.A. Manufacture was banned in the U.S.A. in 1982, and uses ceased in 1986. Toxaphene, the major OC contaminant found in other studies (Muir et al. 1997), was found above the detection levels for only 5 of the 16 fish sampled in this study. The toxaphene levels were again within the range of those reported in Muir et al. (1997) for

Table 16. Coplanar PCB (pg·g^{-1} wet weight) data from lake trout samples (whole fish portions) collected in the Lac de Gras area early in 1994. Sample numbers correspond to those found in Table 14.

Sample Number	33'44'-TPCB PCB 77	344'5-TPCB PCB 81	233'44'-PePCB PCB 105	33'44'5-PePCB PCB 126	33'44'55'-HxPCB PCB 169
1	10.79	1.79	52.75	2.25	5.64
2	24.2	5.65	124.19	3.51	5.91
2*	-	-	-	-	-
3	17.14	5.14	164.46	9.11	13.37
3*	-	-	-	-	-
4	16.03	4.14	142.87	5.34	14.06
5	19.2	1.27	3.91	8.14	18.5
6	8.35	1.41	27.59	5.46	8.24
6*	7.46	1.27	22.72	4.81	8.45
7	38.86	1.87	1.32	3.87	7.69
8	16.38	4.46	111.52	4.08	9.33
8*	-	-	-	-	-
9	20.59	7.41	167.39	6.34	14.41
10	28.8	1.55	3.08	14.51	29.97
11	12.28	3.35	118.07	2.95	8.43
11*	24.62	3.22	73.84	3.67	11.29
12	15.54	0.97	1.52	4.6	9.63
13	16.12	3.86	127.05	4.2	15.1
14	15.64	2.33	37.55	3.38	7.08
15	11.06	0.77	1.71	5.18	7.5
15*	12.76	2.73	48.14	<5.44	<16.84
16	33.87	10.38	798.54	10.11	23.8
16*	-	-	-	-	-

*indicates analysis of duplicate samples

lakes in the Northwest Territories and northern Quebec, but considerably lower than values for Lake Laberge lake trout (Muir et al. 1997). The source of toxaphene in Lake Laberge lake trout has been attributed to long range transport and deposition of organochlorines into the lake and the biomagnification in fish tissue, resulting from an exceptionally long food chain (Kidd et al. 1995). The mean concentrations expressed as ng·g^{-1} wet weight for toxaphene in this study are: Grizzly Lake <30, Long Lake 49.7 ± 31.3, Nero Lake <30, and Vulture Lake 192. For the five fish with detectable levels of toxaphene, the ΣPCB: toxaphene ratio was 2.15, 1.14, 0.96 for Long Lake fish, 0.20 for a fish from Nero Lake, and 0.70 for the fish from Vulture Lake. The ratios for the Long Lake fish are comparable to results for the Lake Laberge fish reported in Muir et al. (1997), where local contamination by PCBs was suspected. Kidd et al. (1995) suggested the use of $\delta^{15}N$ to initially screen for fish with the potential for high organochlorine concentrations. Using this criterion, large Lac de Gras and Long Lake lake trout, which had high $\delta^{15}N$ values, should be screened for organochlorine contaminants.

Polychlorinated dibenzo-p-dioxins (PCDD) and polychlorinated dibenzofurans (PCDF) are large families of chlorinated hydrocarbons, with each congener having a different chlorine substitution pattern. Dioxins and furans share similar physical and chemical properties. They are by-products of industrial processes, principally products of low-temperature, incomplete incineration of chlorine-containing materials, such as plastics. The most toxic of the congeners are substituted at the 2,3,7, and 8 position. Dioxin isomer concentrations (pg·g^{-1} wet weight) are summarized in Table 17. The ΣTCDD concentrations (Table 17) for several individuals were higher than those found in any of the SREQMS individual samples (Sanderson et al. 1997). The ΣH_7CDD from one individual was higher than found in any of the SREQMS individual samples (Sanderson et al. 1997). The remaining congeners were within the range found in individual samples from the SREQMS (Sanderson et al. 1997). Furan isomer concentrations (pg·g^{-1} wet weight) are summarized in Table 18. The concentrations of 1,2,3,4,6,7,8 H_7CDF, and ΣH_7CDF from one individual were higher than found in any of the SREQMS individual samples (Sanderson et al. 1997). The remaining congeners were within the range found in individual samples from the SREQMS (Sanderson et al. 1997). Individual fish data (Table 17 and 18) indicate variability between duplicates and samples. This might have resulted from the use of three different carbon columns for cleanup, the differing rate that the columns became dirty, and the variability between time of cleaning of the columns (Sanderson et al. 1997). The instrument used for dioxin/furan isomer analysis is a high resolution mass spectrometer, which has varying response sensitivities between samples, resulting in the minimum instrument detection limit reported, varying from sample to sample (M. Whittle, Fisheries and Oceans Canada, Bayfield Institute, 867 Lakeshore Rd., Box 5050, Burlington, ON, L7R 4A6, pers. comm.).

Polyaromatic Hydrocarbons (PAH)

PAHs are widely distributed in the environment and are produced both by natural processes (e.g. natural fires and losses from petroleum or coal deposits) and by anthropogenic activities (principally produced during the incomplete combustion of

Table 17. Dioxin isomer (pg g^{-1} wet weight) data from lake trout samples (whole fish portions) collected in the Lac de Gras area early in 1994. Sample numbers correspond to those found in Table 14.

Sample Number	2,3,7,8 TCDD	Σ TCDD	1,2,3,7,8 P_5CDD	Σ P_5CDD	1,2,3,4,7,8 H_6CDD	1,2,3,6,7,8 H_6CDD	1,2,3,7,8,9 H_6CDD	Σ H_6CDD	1,2,3,4,6,7,8 H_7CDD	Σ H_7CDD	OCDD
1	<0.34	<0.34	<0.22	<0.22	<0.06	0.14	<0.05	0.14	0.16	0.16	0.65
2	<0.36	<0.36	<0.11	<0.11	<0.07	0.25	<0.05	0.25	0.21	0.34	0.71
2*	<0.12	9.22	<0.21	<0.21	<0.06	0.11	<0.05	0.11	0.31	0.44	4.33
3	<0.3	4.06	<0.19	<0.19	<0.07	<0.07	0.78	0.99	6.92	7.3	7.59
3*	-	-	-	-	-	-	-	-	-	-	-
4	<0.07	<0.07	<0.14	<0.14	<0.06	<0.04	<0.04	<0.04	0.14	0.23	0.68
5	<2.744	<2.744	<1.082	<1.082	<1.388	<1.134	<1.188	<1.19	<0.734	<0.73	<1.38
6	<0.97	<0.97	<0.933	<0.93	<0.495	<0.405	<0.424	0.66	<1.879	<1.879	8.2
6*	<0.308	<0.31	<0.248	<0.25	<0.091	<0.075	<0.078	<0.08	<0.154	<0.15	0.55
7	<0.288	<0.29	<0.125	<0.13	<0.053	<0.043	<0.045	<0.05	0.2	0.4	2.47
8	<0.05	1.05	<0.09	<0.09	<0.04	<0.03	<0.03	0.18	0.21	0.35	6.78
8*	-	-	-	-	-	-	-	-	-	-	-
9	<0.07	<0.07	<0.14	<0.14	<0.05	<0.04	<0.04	<0.04	0.14	0.14	0.47
10	<0.407	<0.41	<0.284	<0.28	<0.083	<0.068	<0.071	<0.07	0.13	0.13	0.8
11	<0.21	<0.21	<0.15	<0.15	<0.08	<0.06	<0.07	0.12	<0.20	<0.20	6.72
11*	<0.21	<0.21	<0.15	<0.15	<0.07	<0.06	<0.06	<0.06	<0.09	<0.09	0.48
12	<0.43	<0.43	<0.12	<0.12	<0.1	<0.1	<0.09	<0.09	0.18	0.36	0.83
13	<0.72	13.65	<0.33	<0.33	<0.64	<0.9	<0.33	<0.33	4.55	4.55	15.42
14	<0.43	<0.43	<0.13	<0.13	<0.06	<0.04	<0.04	<0.04	<0.05	<0.05	0.63
15	<0.37	<0.37	<0.18	<0.18	<0.08	<0.07	<0.07	<0.07	<0.14	<0.14	0.63
15*	<0.09	6.36	<0.16	<0.16	<0.07	<0.06	<0.05	0.21	0.53	0.63	5.7
16	<0.70	<0.70	<2.29	<2.29	<0.50	<0.41	0.33	1.09	<2.31	<2.31	<1.83
16*	-	-	-	-	-	-	-	-	-	-	-

*indicates analysis of duplicate samples

Σ: values include 2,3,7,8 substituted and non-substituted congeners

Table 18. Furan Isomer ($pg \cdot g^{-1}$ wet weight) data from lake trout samples (whole fish portions) collected in the Lac de Gras area early in 1994. Sample numbers correspond to those found in Table 14.

Sample Number	2,3,7,8 TCDF	Σ TCDF	1,2,3,7,8 P_5CDF	2,3,4,7,8 P_5CDF	Σ P_5CDF	1,2,3,4,7,8 H_6CDF	2,3,4,6,7,8 H_6CDF	1,2,3,6,7,8 H_6CDF	1,2,3,7,8,9 H_6CDF	Σ H_6CDF	1,2,3,4,6,7,8 H_7CDF	Σ H_7CDF	OCDF
1	<0.06	<0.06	<0.14	<0.14	<0.14	<0.08	<0.07	<0.08	<0.09	<0.09	<0.04	<0.04	<0.01
2	0.2	0.2	<0.26	<0.26	<0.26	<0.03	<0.03	<0.03	<0.04	<0.04	0.12	0.12	0.2
2*	<0.13	0.38	<0.19	<0.19	<0.19	<0.05	<0.04	<0.05	<0.06	<0.06	<0.06	<0.06	1.52
3	<0.57	2.55	<0.28	<0.28	<0.28	<0.1	1.06	0.73	0.26	2.05	15.57	18.27	14.06
3*	-	-	-	-	-	-	-	-	-	-	-	-	-
4	<0.2	<0.2	<0.09	<0.09	<0.09	<0.05	<0.04	<0.04	<0.04	<0.04	<0.08	<0.08	<0.08
5	<3.296	<3.3	<0.847	<0.869	<0.87	<0.861	<0.787	<0.857	<0.949	<0.95	<0.149	<0.15	<0.996
6	<0.675	<0.68	<0.573	<0.587	<0.59	<0.357	<0.286	<0.311	<0.344	<0.34	<1.074	<1.07	7.24
6*	<0.10	0.1	<0.137	<0.141	<0.14	<0.065	<0.06	<0.065	<0.072	<0.07	<0.082	<0.08	<0.1
7	<0.14	0.14	<0.07	<0.07	<0.07	<0.04	<0.03	<0.04	<0.04	<0.04	<0.05	<0.05	<0.1
8	<0.20	0.36	<0.11	<0.11	<0.11	<0.03	<0.03	<0.03	<0.03	<0.03	<0.04	<0.04	4.75
8*	-	-	-	-	-	-	-	-	-	-	-	-	-
9	<0.42	0.42	<0.95	<0.097	<0.1	<0.059	<0.061	<0.066	<0.073	<0.07	<0.115	<0.12	<0.1
10	1.01	1.01	<0.17	<0.18	<0.18	<0.06	<0.05	<0.05	<0.06	<0.06	<0.12	<0.12	<0.13
11	0.15	1.91	<0.12	<0.13	0.88	<0.04	<0.04	<0.04	<0.05	<0.05	<0.04	0.18	3.66
11*	0.12	0.17	<0.07	<0.08	<0.08	<0.04	<0.04	<0.05	<0.05	<0.05	<0.12	<0.12	<0.12
12	<1.42	<1.42	<0.13	<0.13	<0.13	<0.1	<0.06	<0.06	<0.07	<0.07	<0.49	<0.49	<0.1
13	<0.87	2.68	<0.55	<0.56	<0.56	<0.46	<0.44	<0.89	<0.33	<0.33	9.91	9.91	20.27
14	<0.27	<0.27	<0.19	<0.19	<0.19	<0.06	<0.04	<0.04	<0.05	<0.05	<0.11	<0.11	0.47
15	<0.24	<0.24	<0.14	<0.15	<0.15	<0.08	<0.05	<0.05	<0.06	<0.06	<0.24	<0.24	<0.18
15*	<0.38	0.47	<0.21	<0.21	<0.21	<0.05	<0.05	<0.05	<0.06	<0.06	1.14	1.14	6.44
16	1.01	1.01	<1.78	<1.82	<1.82	<0.61	<0.66	<0.72	<0.8	<0.8	<1	<1	<1.37
16*	-	-	-	-	-	-	-	-	-	-	-	-	-

*indicates analysis of duplicate samples

Σ: values include 2,3,7,8 substituted and non-substituted congeners

wood, oil, or coal) (Barrie et al. 1997). PAHs are transported to the arctic from temperate industrial regions of the globe in the troposphere, in gas phase, and on particles, as well as via ocean currents (Barrie et al. 1997). This includes long-range transport events, possibly from as far away as western China (Welch et al. 1991). Low $ng \cdot g^{-1}$ concentrations of PAHs were found in fish from the southwestern region of the Northwest Territories, as well as from the Lac de Gras area, analyzed for PAHs as part of the Slave River study and related work (Muir et al. 1997). Data reported in Muir et al. (1997) from Lac de Gras lake trout actually include four lake trout from Lac de Gras and the remaining 14 from Long Lake. Results of the Lac de Gras area sample analysis from the current study (Table 19) are generally quite low. In muscle tissue, low levels, although usually above detection limits, were found for naphthalene, 1- and 2- methylnaphthalene (presented as Σ methylnaphthalene), biphenyl, dibenzofuran, and phenanthrene. Acenaphthylene, acenaphthene, fluorine, and pyrene were usually below detection limits. Anthracene (<0.25), fluoranthene (<0.36), retene (<0.41), benzo(a)anthracene (<0.86), chrysene (<0.71), triphenylene (<0.08), benzo(b)fluoranthene (<1.98), benzo(k)fluoranthene (<0.70), benzo(e)pyrene (<0.87), benzo(a)pyrene (<1.23), indeno(1,2,3-cd)pyrene (<2.54), dibenzo(a,h)anthracene (<1.67), benzo(g,h,i)perylene (<2.57), dibenzothiophene (<0.10), and perylene (<0.81) were all below detection limits. PAH in bile is converted to metabolites that are excreted by the fish. The concentrations of PAH in bile are therefore usually quite low (note: bile concentrations are reported in $pg \cdot g^{-1}$ wet weight of fish tissue), and since the metabolites themselves are not measured, interpretations of PAH concentrations on their own are quite limited. Data presented in Muir et al. (1997) for PAH levels from fish bile are in $ng \cdot mL^{-1}$ of bile (or $ng \cdot g^{-1}$ of bile assuming a bile density of 1). To determine the PAH per gram of fish tissue, the value must be multiplied by the volume of bile (mL), and divided by the wet weight of the fish (g). The resulting concentration is expressed in $pg \cdot g^{-1}$ of fish tissue.

SUMMARY

In 1994, preliminary sampling of fish tissues was undertaken to determine a baseline from which to document any changes in contaminant levels in the diamond mining area near Lac de Gras, NT. From 1994 to 1996, two relatively undisturbed lakes in the area were studied to describe limnological and biological characteristics.

Ice-off occurred around the third week in June. The lakes stratified only temporarily during the open water period in three years of sampling. Summer concentrations of total phosphorous ranged from 4 - 16 $\mu g \cdot L^{-1}$ and total N ranged from 99 - 396 $\mu g \cdot L^{-1}$. Chlorophyll-*a* averaged around 1.3 $\mu g \cdot L^{-1}$ for most of the sampling period for both lakes and all depths. The only exception was following ice-out in Fish2 Lake in 1996 when concentrations averaged 6.7 $\mu g \cdot L^{-1}$ at 1 m and 15.1$\mu g \cdot L^{-1}$ at 4 m water depth. These levels were not matched in Fish1 Lake in 1996, and the period was not sampled during 1995. Zooplankton communities were sampled in both lakes, and the Cyclopoida nauplii were the most important components of the communities, based on numbers of individuals per litre over both years, followed by Calanoida copepodites. Chironomidae emergence began immediately after ice-out reached peak levels shortly after and continued throughout the summer season at a lower level.

Table 19. Polyaromatic Hydrocarbon (PAH) data from fish muscle (ng·g^{-1} wet weight of sample) and bile (pg·g^{-1} wet weight of fish) (± SD) in the Lac de Gras area in 1994.

Species	Lake	Tissue	N	Naphthalene	Σmethyl[1] naphthalene	Biphenyl	Dibenzofuran	Acenaphthylene	Fluorene	Phenanthrene	Anthracene	Fluoranthene	Pyrene	total PAH[2]
arctic grayling	Fish1 (Cam)	M	8	2.44±0.72	3.28±1.32	0.48±0.18	0.17±0.03	0.05±0.05	0.04±0.07	0.28±0.05	0.07±0.05	0.02±0.03	0.02±0.04	6.19±2.03
		B	3	34.2±10.2	99. 5±110.8	3.0±0.9	0.9±1.6	0	1.3±2.3	4.1±1.1	0.6±1.1	0	0	401.7±248.6
	Fish 2 (Ron)	M	6	2.65±1.11	3.01±1.22	0.53±0.23	0.13±0.05	0.09±0.17	0.09±0.08	0.27±0.06	0.04±0.05	0.07±0.04	0.10±0.06	6.32±2.64
		B	1	11.6	11.3	-	0	0	1.1	1.1	0	0	0	50.4
lake trout	Fish1 (Cam)	M	11	2.58±0.80	3.92±1.47	0.54±0.13	0.17±0.03	0.07±0.07	0.17±0.08	0.27±0.06	0.03±0.04	0.02±0.03	0.02±0.03	7.08±2.24
		B	5	32.7±9.5	121.6±138.1	2.2±2.0	2.5±1.7	0	0.3±0.6	4.6±2.4	0.2±0.4	0	1.7±2.6	465.1±441.9
	Fish 2 (Ron)	M	10	1.78±0.30	2.22±0.39	0.47±0.13	0.15±0.04	0.00±0.00	0.05±0.07	0.24±0.05	0.03±0.04	0.01±0.03	0.05±0.06	4.39±0.73
		B	6	13.3±3.6	13.4±3.1	-	0.4±0.6	0	0.1±0.1	1.8±1.0	0	0.9±2.2	6.2±15.0	69.2±31.5
	Lac de Gras	M	4	4.27±1.59	1.90±2.33	0.45±0.24	0.18±0.11	0.02±0.04	0.08±0.13	0.24±0.13	0.04±0.04	0.09±0.06	0.12±0.09	6.74±3.01
	Long	M	14	4.81±2.51	1.91±2.68	0.46±0.18	0.22±0.11	0.02±0.06	0.10±0.33	0.39±0.25	0.03±0.04	0.06±0.05	0.08±0.06	7.41±3.57
round whitefish	Fish1 (Cam)	M	10	2.72±0.92	4.19±1.40	0.59±0.20	0.18±0.03	0.02±0.04	0.06±0.07	0.22±0.04	0.03±0.05	0.01±0.02	0.03±0.05	7.27±2.30
	Fish 2 (Ron)	M	11	2.12±0.62	2.32±0.61	0.42±0.16	0.17±0.06	0.07±0.08	0.09±0.11	0.19±0.02	0.04±0.03	0.05±0.05	0.07±0.05	4.95±1.31
		B	5	11.2±5.0	36.6±59.7	-	0.4±0.7	0	0.1±0.2	1.4±0.7	0.1±0.3	0.2±0.4	0.3±0.6	91.9±117.2
	Long	M	22	3.91±1.17	3.86±3.18	0.65±0.29	0.29±0.10	0.03±0.04	0.03±0.06	0.29±0.09	0.04±0.03	0.06±0.05	0.17±0.12	8.38±3.73

[1] sum of 1- and 2- methyl naphthalene

[2] total PAH is the sum of 16 PAHs identified as priority pollutants plus 1- and 2-methyl naphthalene (total PAH does not including perylene and retene)

Both lakes have populations of lake trout, round whitefish, arctic grayling, and burbot. Cyprinidae and sculpins were present in the lakes, but there was no attempt to sample them. Little information was collected for burbot. Round whitefish dominated the fish community in both lakes. Fish2 Lake supports a larger population of lake trout than arctic grayling, which is the opposite of the community structure in Fish1 Lake. Both round whitefish and arctic grayling populations were dominated by smaller size classes. Larger individuals dominated the lake trout populations. The diets differed between species, although zooplankton was the major diet item consumed by all species. Lake trout also fed on Chironomidae and fish. Other prey items included several species of aquatic insects, molluscs, Notostraca, and terrestrial insects. Arctic grayling also consumed Chironomidae, and to a lesser extent, several aquatic insects, terrestrial insects, and Notostraca. Round whitefish relied less on zooplankton than the other two species, and consumed proportionally more Chironomidae, Notostraca, Trichoptera, and Mollusca. Within a given lake, arctic grayling were positioned at the lowest trophic level, and lake trout at the highest. Round whitefish were intermediate between the two, although they tended to be less depleted in ^{13}C, likely as a result of the greater importance of benthic invertebrates and the lower reliance on zooplankton for this species compared to the other two. Streams were briefly surveyed, and were used by arctic grayling, lake trout, burbot and sculpin. They allowed access between the two lakes.

High levels of Cu found in liver tissue suggest exposure to this metal. Levels of Se in both muscle and liver tissue are near or at the toxic effect thresholds for overall health and reproductive vigor in fish. Hg was above the guidelines for subsistence fish consumption in the larger lake trout sampled. All other metals analyzed are within the range found in other studies in the Northwest Territories. MFO activity, particularly as AHH, was higher in individual fish than found in most other arctic samples. These levels may expand the range of normal responses for arctic fish or may indicate exposure to some type of contaminant. OC contaminants in general were found at low levels, although total PCB levels were higher than toxaphene in the four lakes sampled. PAH values were low. At the time of this study, there were no apparent differences in the contaminant levels in fish from the two drainage basins sampled.

Continued monitoring of the contaminant levels of fish from lakes in the area should be carried out. Metals in particular should be monitored to ascertain changes resulting from the redistribution of elements as a consequence of ongoing mining operations. Gills should be evaluated for metal content and histological alteration. Contaminant analysis should be conducted on burbot. Liver, kidney, and muscle should be analyzed for all fish. All analyses should be conducted on the same fish samples to determine if relationships exist between contaminants and biomarkers. Sampling done, as part of this study, was not sufficient to allow for an accurate assessment of the fish populations. Assessment of the Cyprinidae, and burbot populations is needed to understand the entire ecosystem. Further work is also needed to determine the importance of the stream habitats, to the fish populations.

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REFERENCES

Arnason, A.N., C.J. Schwarz, and G. Boyer. 1998. Popan-5: a data maintenance and analysis system for mark-recapture data. Scientific Report, Department of Computer Science, University of Manitoba, Winnipeg. 265 p.

Barrie, L., R. Macdonald, T. Bidleman, M. Diamond, D. Gregor, R. Semkin, W. Strachan, M. Alaee, S. Backus, M. Bewers, C. Gobeil, C. Halsall, J. Hoff, A. Li, L. Lockhart, D. Mackay, D. Muir, J. Pudykiewicz, K. Reimer, J. Smith, G. Stern, W. Schroeder, R. Wagemann, F. Wania, and M. Yunker. 1997. Chapter 2. Sources, Occurrence and Pathways, p. 25-182. *In* Jensen, J., K. Adare, and R. Shearer (ed.) Canadian Arctic Contaminants Assessment Report, Indian and Northern Affairs Canada, Ottawa 1997. 460 p.

Beamish, R.J. 1972. Design of a trapnet for sampling shallow-water habitats. Fish. Res. Board Can. Tech. Rep. 305: 14 p.

Boyle, D.E., B.A. Bravender, T.J. Brown, D. Kieser, C.D. Levings, W.L. Lockhart, J.A. Servizi, and T.R. Whitehouse. 1992. Baseline monitoring of mountain whitefish *Prosopium williamsoni*, from the Columbia River system near Castlegar, British Columbia: health, contaminants and biology. Can. Tech. Rep. Fish. Aquat. Sci. 1883: x + 120 p.

Cabana, G., A. Tremblay, J. Kalff, and J.B. Rasmussen. 1994. Pelagic food chain structure in Ontario lakes: a determinant of mercury levels in lake trout (*Salvelinus namaycush*). Can. J. Fish. Aquat. Sci. 51: 381-389.

Davies, I.J. 1984. Sampling aquatic insect emergence, p. 161-227. *In* J.A. Downing and F.H. Rigler (ed.) A manual on methods for the assessment of secondary productivity in fresh waters. IBP Handbook 17, Blackwell Scientific Publications, Oxford. 501 p.

de March, B.G., C.A. de Wit, D.C. Muir, B.M. Braune, D.J. Gregor, R.J. Norstrom, M. Olsson, J.U. Skaare, and K. Stange. 1998. Chapter 6 Persistent Organic Pollutants, p. 183-371. *In* de March, B.G., C.A. de Wit, and D.C. Muir (ed.) AMAP Assessment Report: Arctic Pollution Issues. 1998. Arctic Monitoring and Assessment Programme (AMAP), Oslo, Norway. 859 p.

Depierre, J.W., Moron, M.S., K.A.M. Johannesen, and L. Ernster. 1975. A reliable, sensitive and convenient radioactive assay for benzpyrene monooxygenase. Anal. Biochem. 63: 470-484.

Förstner, U., and G.T.W. Wittmann (ed.) 1981. Metal pollution in the aquatic environment. Second Revised Edition. Springer-Verlag, New York. 486 p.

George, E.L., and W.F. Hadley. 1979. Food and habitat partitioning between rock bass (*Ambloplites rupestris*) and smallmouth bass (*Micropterus dolomieui*) young of the year. Trans. Am. Fish. Soc. 108: 253-261.

Harper, F. 1948. Fishes of the Nueltin Lake expedition, Keewatin, 1947, Pt. 2 – Historical and field notes. Proc. Acad. Nat. Sci. Phil. 100: 153-184.

Harrison, S.E., R.V. Hunt, R.E. Evans, R.E. McNicol, and J.F. Klaverkamp. 1995. A preliminary evaluation of the lethal and sub-lethal toxicities to fish of effluents from diamond exploration in the Lac de Gras Area. Environment Canada, Environmental Protection Branch, Yellowknife, NWT. EC/EPB/TS-32. 52 p.

Health and Welfare Canada. 1984. Methylmercury in Canada. Exposure of Indian and Inuit residents to methylmercury in the Canadian environment. Ottawa: Health and Welfare Canada, Medical Services Branch. 164 p.

Hendzel, M.R., and D.M. Jamieson. 1976. Determination of mercury in fish. Anal. Chem. 48: 926-928.

Huestis, S.Y., and D.B. Sergeant. 1992. Removal of chlorinated diphenyl ether interferences for analyses of PCDDs and PCDFs in fish. Chemosphere 24: 537-545.

Huestis, S.Y., M.R. Servos, D.B. Sergeant, M. Leggett, and D.G. Dixon. 1995. Methods for determination of organochlorine pesticides, polychlorinated biphenyl congeners and chlorinated dibenzo-p-dioxins and furans in fish. Can. Tech. Rep. Fish. Aquat. Sci. 2044: iv + 26 p.

Jensen, J., K. Adare, and R. Shearer (ed.) 1997. Canadian Arctic Contaminants Assessment Report. Indian and Northern Affairs Canada, Ottawa. 460 p.

Jolly, G.M. 1965. Explicit estimates from capture-recapture data with both death and immigration – stochastic model. Biometrika 52:225-247.

Jolly, G.M. 1982. Mark-recapture models with parameters constant in time. Biometrics 38: 301-321.

Jolly, G.M., and J. Dickson. 1980. Mark-recapture suite of programs. p. 570-576. *In* Barritt, M.M., and D. Wishart (ed.) Compstat-1980: proceedings in computational statistics. Vol 4. Vienna: Physica-Verlag.

Kidd, K.A., M.J. Paterson, R.H. Hesslein, D.C.G. Muir, and R.E. Hecky. 1999. Effects of northern pike (*Esox lucius*) additions on pollutant accumulation and food web structure, as determined by $\delta^{13}C$ and $\delta^{15}N$, in a eutrophic and an oligotrophic lake. Can. J. Fish. Aquat. Sci. 56: 2193-2202.

Kidd, K.A., D.W. Schindler, R.H. Hesslein, and D.C.G. Muir. 1998. Effects of trophic position and lipid on organochlorine concentrations in fishes from subarctic lakes in Yukon Territory. Can. J. Fish. Aquat. Sci. 55: 869-881.

Kidd, K.A., D.W. Schindler, D.C.G. Muir, W.L. Lockhart, and R.H. Hesslein. 1995. High concentrations of toxaphene in fishes from a subarctic lake. Science 269: 240-242.

Köck, G., M. Triendl, and R. Hofer. 1996. Seasonal patterns of metal accumulation in arctic char (*Salvelinus alpinus*) from an oligotrophic alpine lake related to temperature. Can. J. Fish. Aquat. Sci. 53: 780-786.

Lemly, A.D. 1996. Selenium in Aquatic Organisms. Chapter 19. *In*: Beyer, W.N, G.H. Heinz and A.W. Redmon-Norwood (ed.) Environmental Contaminants in Wildlife: Interpreting Tissue Concentrations. SETAC Special Publications Series. Lewis Publishers, Boca Raton, Florida 494 p.

Lockhart, W.L. 1995. Implications of chemical contaminants for aquatic animals in the Canadian Arctic: some review comments. Sci. Total Environ. 160/161: 631-641.

Lockhart, W.L., D.A. Metner, D.A.J. Murray, R.W. Danell, B.N. Billeck, C.L. Baron, D.C.G. Muir, and K. Chang-Kue. 1989. Studies to determine whether the condition of fish from the lower Mackenzie River is related to hydrocarbon exposure. Department of Indian and Northern Affairs, Northern Oil & Gas Action Program (NOGAP). Environmental Studies No. 61. 84 p.

McPhail, J.D., and C.C. Lindsey. 1970. Freshwater fishes of northwestern Canada and Alaska. Bull. Fish. Res. Board Can. 173: 381 p.

Muir, D., B. Braune, B. deMarch, R. Norstrom, R. Wagemann, M. Gamberg, K. Poole, R. Addison, D. Bright, M. Dodd, W. Duschenko, J. Eamer, M. Evans, B. Elkin, S. Grundy, B. Hargrave, C. Hebert, R. Johnstone, K. Kidd, B. Koenig, L. Lockhart, J. Payne, J. Peddle, and K. Reimer. 1997. Chapter 3. Ecosystem uptake and effects, p. 183-294. *In* Jensen, J., K. Adare, and R. Shearer (ed.) Canadian Arctic Contaminants Assessment Report. Indian and Northern Affairs Canada, Ottawa 1997. 460 p.

Omura, T., and R. Sato. 1964a. The carbon monoxide-binding pigment of liver microsomes I. Evidence for its hemo-protein nature. J. Biol. Chem. 239: 2370-2378.

Omura, T., and R. Sato. 1964b. The carbon monoxide-binding pigment of liver microsomes II. Solubilization, purification, and properties. J. Biol. Chem. 239: 2379-2385.

Parker, H.H., and L. Johnson. 1991. Population structure, ecological segregation and reproduction in non-anadromous arctic charr, *Salvelinus alpinus* (L.), in four unexploited lakes in the Canadian high Arctic. J. Fish Biol. 38: 123-147.

Pohl , R.J. and J.R. Fouts. 1980. A rapid method for assaying the metabolism of 7-ethoxyresorufin by microsomal cellular fractions. Anal. Biochem. 107: 150-155.

Rawson, D.S. 1950. The grayling (*Thymallus signifer*) in northern Saskatchewan. Can. Fish Culturist 6: 3-10.

Rawson, D.S. 1951. Studies of the fish of Great Slave Lake. J. Fish. Res. Board Can. 8: 207-240.

Reinke, J., J.F. Uthe, and T. Sabatino. 1972. Organochlorine residues in commercial fish caught in Canada – 1970. Pestic. Monit. J. 6: 43-49.

Rescan Environmental Services Ltd. (Rescan). 1993a. Bulk sampling project description. Report submitted to Regional Environmental Review Committee, INAC, Yellowknife by BHP Minerals Canada Ltd.

Rescan Environmental Services Ltd. 1993b. Lake bathymetry and morphometry. Report submitted to BHP Minerals Canada Ltd.

Rescan Environmental Services Ltd. 1993c. Baseline environmental study protocols. Report submitted to Regional Environmental Review Committee, INAC, Yellowknife by BHP Minerals Canada Ltd.

Rescan Environmental Services Ltd. 1994a. Project description report. Report submitted to Regional Environmental Review Committee, INAC, Yellowknife by BHP Minerals Canada Ltd.

Rescan Environmental Services Ltd. 1994b. Panda Lake diversion and fish habitat enhancement channel design. Report submitted to BHP Minerals Canada Ltd.

Rescan Environmental Services Ltd. 1995. Towards a fisheries compensation plan, estimation of fish habitat in Koala District lakes and streams. Report submitted to BHP Minerals Canada Ltd.

Rescan Environmental Services Ltd. 1996. Sable Lake 1996 Environmental data report - primary and secondary producers. Report prepared for BHP Diamonds Inc.

Rigler, F.H. 1972. The Char Lake project. A study of energy flow in a high arctic lake, p. 278-198. *In* Z. Kajak and A. Hillbricht-Ilkowska (ed.) Productivity problems in freshwater. PWN, Warszawa

Rigler, F.H. 1975. The Char Lake project. An introduction to limnology in the Canadian Arctic, p. 171-198. *In* T.W. Cameron and L.W. Billinglsley (ed.) Energy flow: its biological dimensions: a summary of the IBP in Canada, 1964 -1974. Royal Soc. of Canada, Ottawa.

Sanderson (Peddle), J., C. Lafontaine and K. Robertson. 1997. Slave River Environmental Quality Monitoring Program Final Five Year Study Report, 1990-95. Water Resource Division, Indian and Northern Affairs Canada, Yellowknife, NWT. 3 Vols.

Scott, W.B., and E.J. Crossman. 1973. Freshwater fishes of Canada. Fish. Res. Bd. Canada, Bull. 184: 965 p.

Seber, G.A.F. 1982. The estimation of animal abundance and related parameters. 2nd ed. Macmillan Publishing Co. Inc., New York, NY. 654 p.

Smith, T.G., and F.A.J. Armstrong. 1975. Mercury in seals, terrestrial carnivores and principal food items of the Inuit, from Holman, N.W.T. J. Fish. Res. board Can. 32: 795-801.

Sotiropoulos, M. 1997. The food habits of fish found in a freshwater arctic ecosystem. B.Sc.Thesis, Univ. Manitoba, Dept. Zool., Winnipeg, MB. Xxp.

Stainton, M.P., M.J. Capel, and F.A.J. Armstrong. 1977. The chemical analysis of freshwater. 2nd ed. Can. Fish. Mar. Serv. Misc. Spec. Publ. 25: vi + 180 p.

Van Cantfort, J., J. Degraeve and J.E. Gielen. 1977. Radioactive assay for aryl hydrocarbon hydroxylase. Improved method and biological importance. Biochem. Biophys. Res. Commun. 79: 505-512.

Vijan, P.N., and G.R. Wood. 1974. An automated submicrogram determination of arsenic in atmospheric particulate matter by flameless atomic absorption spectrophotometry. A.A. Newsletter 13: 33-37.

Welch, H.E. 1985. Introduction to limnological research at Saqvaqjuac, Northern Hudson Bay. Can. J. Fish. Aquat. Sci. 42: 494-505.

Welch, H.E., and M.A. Bergmann. 1985. Winter respiration of lakes at Saqvaqjuac, N.W.T. Can. J. Fish. Aquat. Sci. 42: 521-528.

Welch, H.E., and K.H. Mills. 1981. Marking fish by scarring soft fin rays. Can. J. Fish. Aquat. Sci. 38: 1168-1170.

Welch, H.E., J.K. Jorgenson, and M.F. Curtis. 1988. Emergence of Chironomidae (Diptera) in fertilized and natural lakes at Saqvaqjuac, N.W.T. Can. J. Fish. Aquat. Sci. 45: 731-737.

Welch, H.E., J.A. Legault, and H.J. Kling. 1989. Phytoplankton, nutrients, and primary production in fertilized and natural lakes at Saqvaqjuac, N.W.T. Can. J. Fish. Aquat. Sci. 46: 90-107.

Welch, H.E., D.C.G. Muir, B.N. Billeck, W.L. Lockhart, G.J. Brunskill, H.J. Kling, M.P. Olson, and R.M. Lemoine. 1991. Brown snow: a long-range transport event in the Canadian arctic. Environ. Sci. Tech. 25: 280-286.

Williams, T.G., W.L. Lockhart, D.A. Metner and S. Harbicht. 1997. Baseline studies in the Slave River, NWT, 1990-1994: Part III. MFO enzyme activity in fish. Sci. Total Environ. 197: 87-109.

Wilkinson, L., J. Harris, B. Kjarsgaard, B. McClenaghan and D. Kerr. 2001. Influence of till thickness and texture on till geochemistry in the Lac de Gras area, Northwest Territories, with applications for regional kimberlite exploration. Geological Survey of Canada, Current Research 2001-C9, 17 p.

APPENDIX 1. Photographs from Fish2 Lake, 1996.

Fish2 Lake viewed from above the point on the west shore towards the south end of the lake. Fish1 Lake is visible in the distance. Photograph by M. Sotiropoulos.

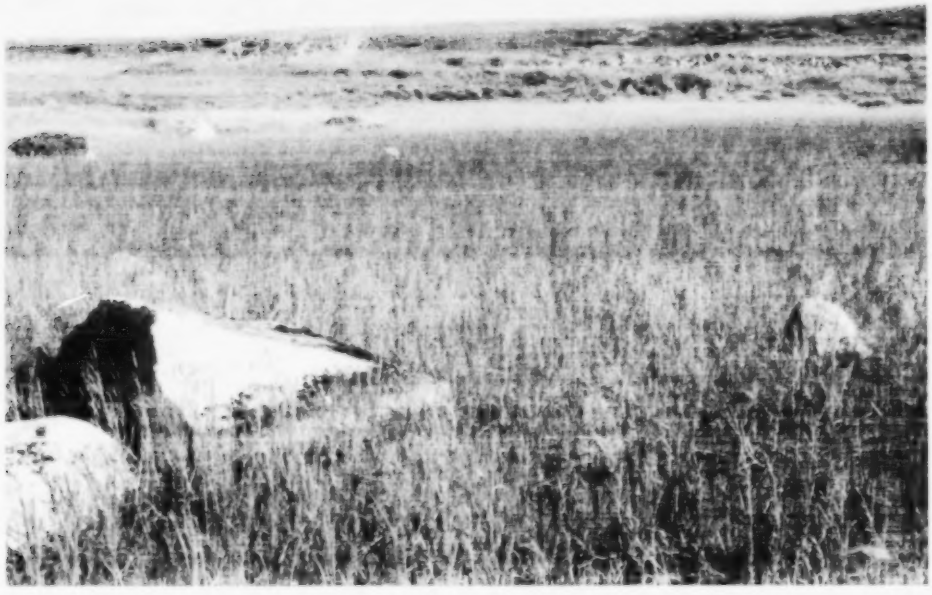
APPENDIX 1. (Continued) Photographs from Fish2 Lake, 1996.

The southwest shoreline of Fish2 Lake viewed from the point in the previous photograph. Photograph by M. Sotiropoulos.

APPENDIX 1. (Continued) Photographs from Fish2 Lake, 1996.

View of the point on the western shore near the south end of Fish2 Lake. Emergent vegetation is visible extending out into the lake. Photograph by M. Sotiropoulos.

APPENDIX 1. (Continued) Photographs from Fish2 Lake. 1996.



The shore near the north end of Fish2 Lake with dense emergent vegetation. The ridge in the distance is an esker which was followed en route to Fish1 Lake. Photograph by M. Sotiropoulos.

APPENDIX 1. (Continued) Photographs from Fish2 Lake, 1996.

Fish2 Lake inflow stream from Rae Lake. Photograph by M. Sotiropoulos.

APPENDIX 1. (Continued) Photographs from Fish2 Lake, 1996.

Boulders with periphyton in the stream entering Fish2 Lake.
Photograph by M. Sotiropoulos.